National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Nearshore Marine Vital Signs Monitoring in the Southwest Alaska Network of National Parks

2010

Natural Resource Technical Report NPS/SWAN/NRTR—2011/497

ON THE COVER
Kenai Fjords National Park
Photograph by: Tjibbe Stelwagen, USGS volunteer

Nearshore Marine Vital Signs Monitoring in the Southwest Alaska Network of National Parks

2010

Natural Resource Technical Report NPS/SWAN/NRTR—2011/497

Heather A. Coletti

National Park Service
240 W 5th Avenue
Anchorage, AK 99501

James L. Bodkin

US Geological Survey
Alaska Science Center
4210 University Dr
Anchorage, AK 99508

Thomas A. Dean

Coastal Resources Associates Inc.
302a Pier View Way
Oceanside, CA 92054

Kimberly A. Kloecker

US Geological Survey
Alaska Science Center
4210 University Dr
Anchorage, AK 99508

October 2011

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado publishes a range of reports that address natural resource topics of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

The Natural Resource Technical Report Series is used to disseminate results of scientific studies in the physical, biological, and social sciences for both the advancement of science and the achievement of the National Park Service mission. The series provides contributors with a forum for displaying comprehensive data that are often deleted from journals because of page limitations.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner.

This report received informal peer review by subject-matter experts who were not directly involved in the collection, analysis, or reporting of the data.

This report is available from (http://science.nature.nps.gov/im/units/swan/) and the Natural Resource Publications Management website (http://www.nature.nps.gov/publications/nrpm/).

Please cite this publication as:

Coletti, H. A., J. L. Bodkin, T. A. Dean, and K. A. Kloecker. 2011. Nearshore marine vital signs monitoring in the Southwest Alaska Network of National Parks: 2010. Natural Resource Technical Report NPS/SWAN/NRTR—2011/497. National Park Service, Fort Collins, Colorado.

NPS 965/110809, October 2011

Contents

Contents (continued)

Figures

Figures (continued)

Page

Figures (continued)

Page

Tables

Page

Abstract

In 2010, we successfully completed a fifth year of protocol testing and field sampling for the Southwest Alaska Network's (SWAN) Nearshore Vital Signs monitoring program in accordance with standard operating procedures set forth for each of six vital signs: marine intertidal invertebrates, kelp and seagrass, marine water chemistry and quality, marine birds, black oystercatcher, and sea otter.

Summer sampling in 2010 represented the fifth year of data collection at Katmai National Park & Preserve (KATM) and fourth year of data collection in Kenai Fjords National Park (KEFJ) for vital signs including intertidal invertebrates and algae, water chemistry and quality, marine bird surveys, black oystercatcher diet and productivity, and sea otter diet. No modifications were made to the rocky intertidal sampling protocol from previous years and the protocol and SOPs have been finalized. Hobo water temperature sensors are currently deployed at five rocky intertidal sites in both KATM and KEFJ. In addition, salinity loggers were co-located at all rocky intertidal sites at both KATM and KEFJ in 2010. We implemented a third year of mussel bed and eelgrass bed sampling at both parks. A final SOP for sampling mussel beds is complete and is currently out for peer review. A final SOP for eelgrass bed monitoring is in final draft form and will be sent for review in the fall of 2011.

Marine bird surveys in KEFJ will continue with little modification in 2011. KATM will not be sampled in 2011, but will be again in 2012. For marine bird surveys, we recommend that current survey effort continue until further analysis can be completed. The existing SOP for marine bird surveys is final. While not reported here, a winter marine bird survey was conducted in March of 2010 in KEFJ. (http://science.nature.nps.gov/im/units/swan/Libraries/Reports/ColettiH_2010_KEFJ_WinterMarineBirdMammalDistributionDensityNRTR_2170928.pdf)
Black oystercatcher abundance, nests density, productivity and diet data should continue to be collected with little revision. Sampling at the current intensity should allow us to detect trends in changes of nest density, productivity and diet (especially prey size) of the black oystercatcher. The SOP for black oystercatcher monitoring is also final. Although an aerial survey of sea otter abundance was completed in KEFJ during June 2010, results from the survey are reported elsewhere and available. (http://science.nature.nps.gov/im/units/swan/Libraries/Reports/ColettiH_2011_KEFJ_SeotAerial2010Report_2167598.pdf)
Sea otter foraging data was collected at both parks. In general, mussels continue to be the dominate prey in KEFJ and clams the dominate prey in KATM. A sea otter forage database has been completed. Database completion will ease data entry both in the field and office as well as optimize data analysis. Carcass collection continues in KATM and KEFJ, although to date we have not recovered enough carcasses from KEFJ to employ age-specific mortality analyses.

In 2011, we plan to finalize the protocol narrative through an external peer review process and finalize data entry and data management procedures for the rocky intertidal SOP. We will continue to sample nearshore vital signs at KEFJ in 2011 and return to sampling both parks in 2012.

Acknowledgments

The National Park Service, SWAN, KATM, KEFJ, LACL and the USGS Alaska Science Center supported this work. We would like to recognize the exceptional cooperation by the staff of KATM, KEFJ and SWAN, in particular; Laura Phillips and Mark Kansteiner (KEFJ) and Carissa Turner (KATM) for their field assistance in 2010. This work could not have been completed without the field assistance of Allan Fukuyama (contractor), George Esslinger (USGS), Sue Saupe (CIRCAC), Robin Corocoran (USFWS) Vanessa VonBiela (USGS), and a great group of volunteers including: Kelly Bodkin, Chris Ehrler, Jessica Perry and Tjibbe Stelwagen. We would like to thank Bill Thompson, Michael Shephard, Cuyler Smith and Claudette Moore of SWAN NPS for their continued technical support. Thank you to Claudette Moore and James Walton for their thoughtful reviews.

We also want to extend a special 'thank you' to Greg Snedgen and George Esslinger for their skilled operation of the R/V Alaskan Gyre in KEFJ and KATM.

Intertidal Invertebrates and Algae

Introduction

Intertidal invertebrate and algal communities provide an important source of production; are an important conduit of energy, nutrients, and pollutants between terrestrial and marine environments; provide resources for subsistence, sport, and commercial harvests; and are important for recreational activities such as wildlife viewing and fishing. The intertidal is particularly susceptible to human disturbance including oil spills; trampling by recreational visitors; harvesting activities; pollutants from terrestrial, airborne and marine sources; and shoreline development. Changes in the structure of the intertidal community serve as valuable indicators of disturbance, both natural (e.g. Dayton 1971, Sousa 1979) and human induced (Barry et al. 1995, Lewis 1996, Keough and Quinn 1998, Jamieson et al. 1998, Shiel and Taylor 1999, Sagarin et al. 1999, Peterson 2001, and Peterson et al. 2003).

Intertidal invertebrates and algae (including intertidal kelps) were sampled annually at KATM beginning in 2006, and at KEFJ beginning in 2008. Sampling of intertidal invertebrates and algae at these sites is designed to detect changes in these communities over time as part of the SWAN Vital Signs program. The specific objectives of this sampling on rocky shores are to assess changes in: 1) the relative abundance of algae, sessile invertebrates, and motile invertebrates in the intertidal zone, 2) the diversity of algae and invertebrates 3) the size distribution of limpets (*Lottia persona*) and mussels (*Mytilus trossulus*), 4) the concentration of contaminants in mussel tissue, and 5) temperature (either sea or air depending on tidal stage). In this section, we present results of sampling conducted in 2010. The metrics to be examined are: 1) abundance estimates for dominant taxa of sessile invertebrates and algae and the size distribution of the limpet *Lottia persona*.

Methods

Sampling was conducted at five sites in sheltered rocky habitats within KATM and KEFJ in 2010. Descriptions of the study sites and methods used to sample intertidal algae and invertebrates are available in Dean and Bodkin (2011b). The following is a general description of the methods employed. Sampling of abundance and species composition for algae and invertebrates was conducted along two 50 m linear transects at each site. The percent cover of algae and sessile invertebrates was estimated within 12 evenly spaced ¼ m^2 quadrats placed along transects that ran parallel to the shoreline and originated at permanent markers placed at 0.5 m and 1.5 m tidal elevations, respectively. Quadrats were placed at a random start points and at equally spaced intervals thereafter. In addition, a minimum of 119 individual limpets (*Lottia persona*) were measured at each site for estimation of size distributions.

The analyses presented here focus on estimates of abundance of dominant taxa at each tidal elevation, and on size distributions of limpets. Means and 95% confidence intervals are reported for each park in each year.

Results

Mean percent cover (and 95% confidence intervals) are reported for each site at KATM and KEFJ in Figures 1 through 11. Relative abundance varied by park and tidal elevation, but *Fucus distichus subsp. evanescens,* barnacles, and *Alaria marginata* (at 0.5 m mean lower low water

(MLLW) in KATM) were generally the most abundant. Notable differences between parks were observed at the lower (0.5 m MLLW) tidal elevation, with a greater percent cover of *Fucus* at KEFJ and greater cover by *Alaria* and barnacles at KATM. The only notable trend over time was an increase in cover by *Fucus* at both 0.5 and 1.5 m tidal elevations at KEFJ between 2008 and 2010. The mean abundance of *Fucus* at both tidal elevations at KEFJ sites more than doubled over this time. No differences between parks or years were noted for the mean size of the limpet *Lottia persona*.

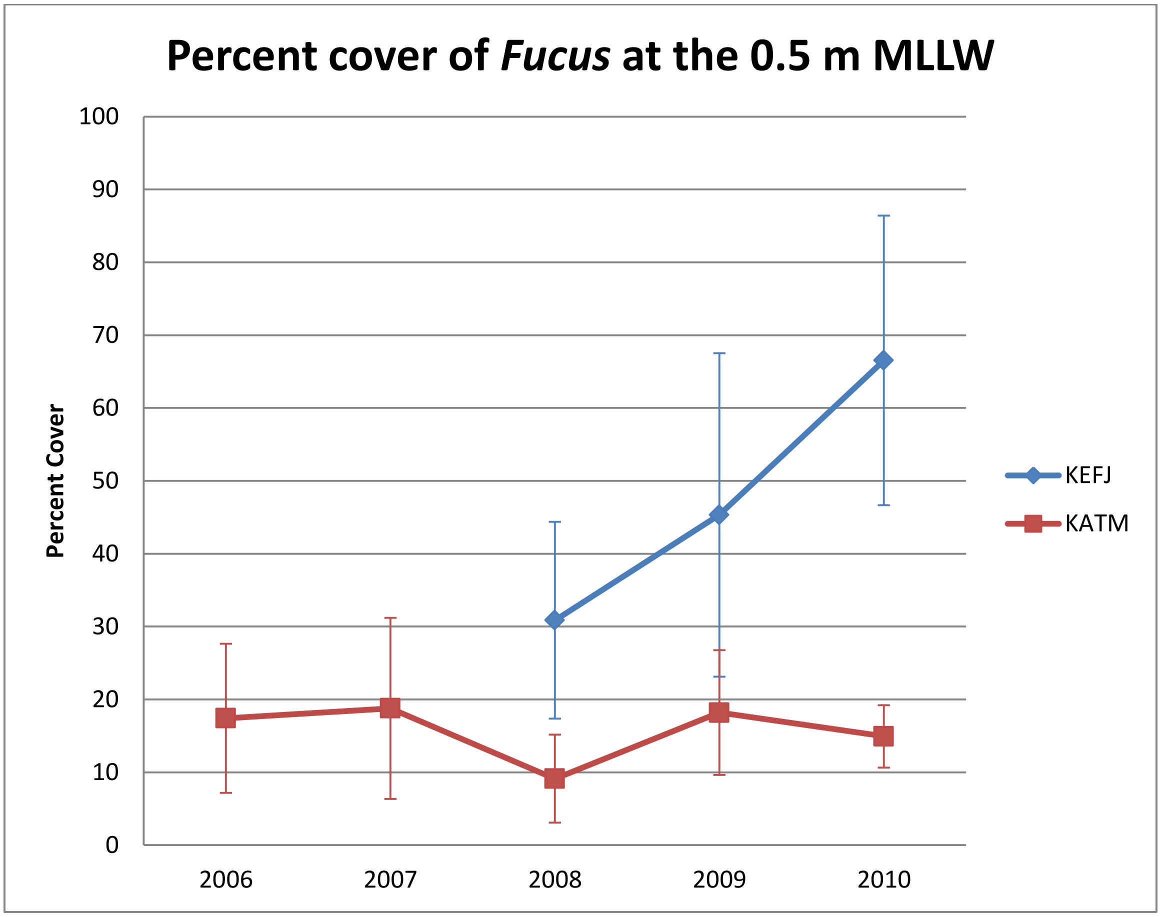


Figure 1. Percent cover of *Fucus* at the 0.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

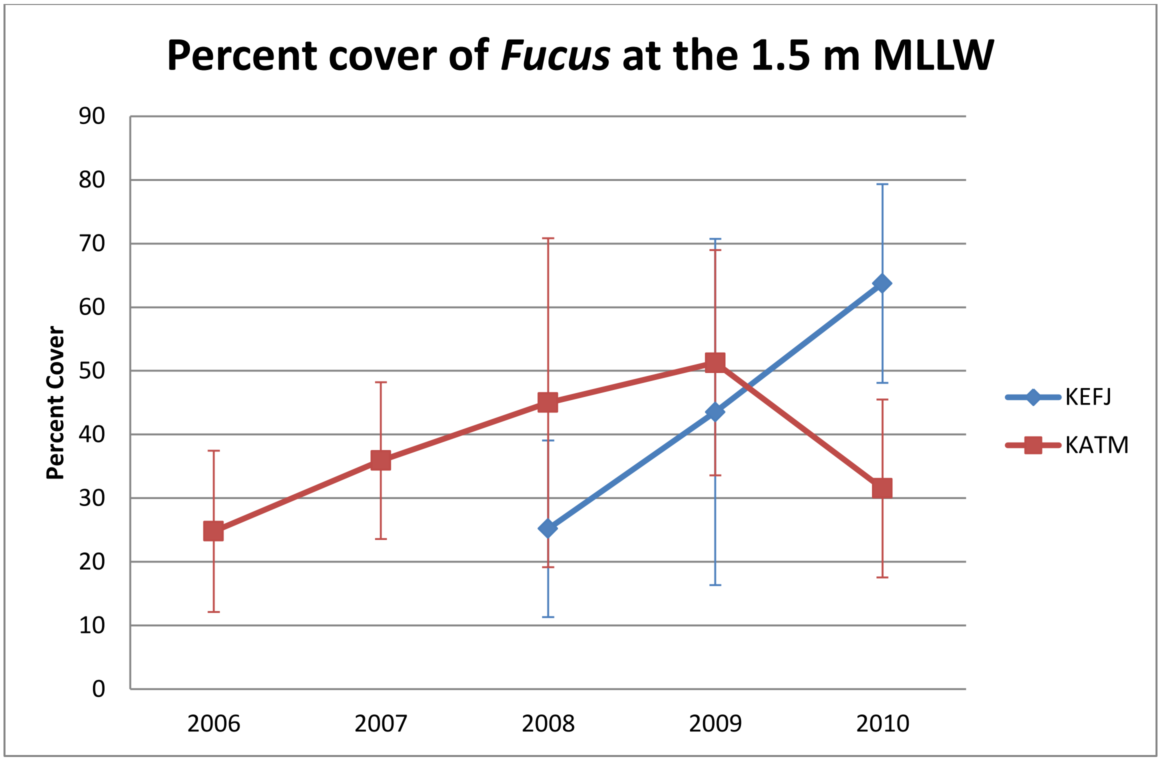


Figure 2. Percent cover of *Fucus* at the 1.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

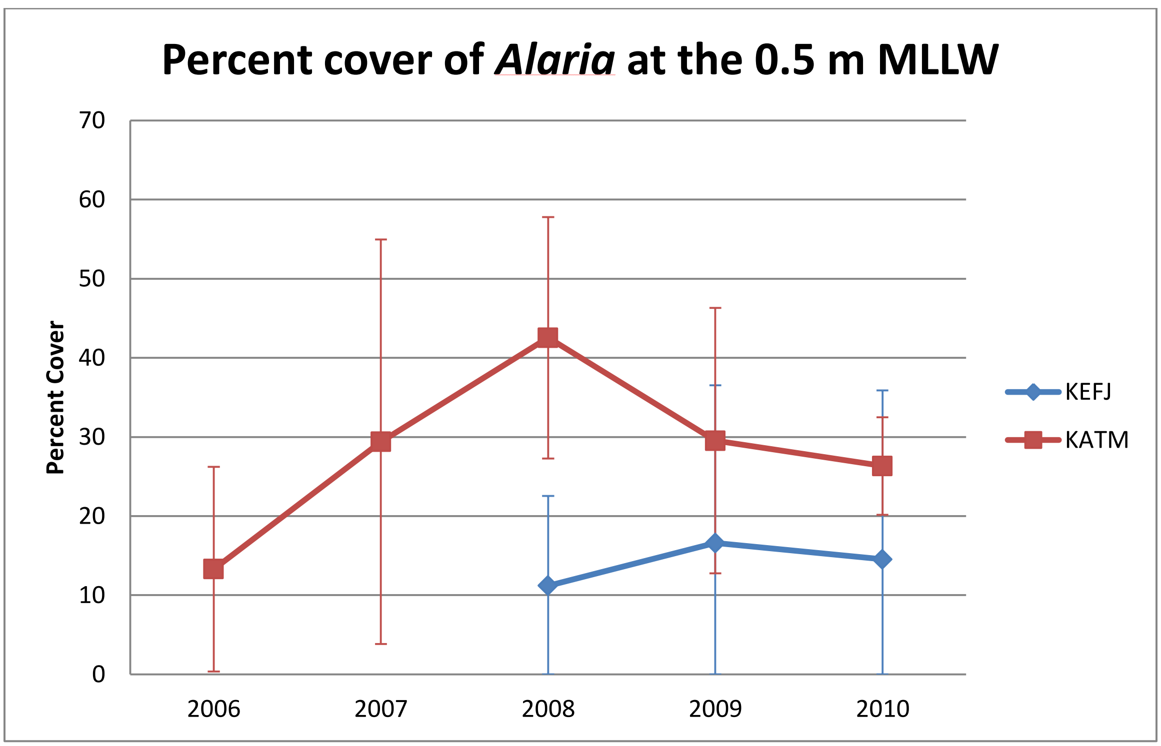


Figure 3. Percent cover of *Alaria* at the 0.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

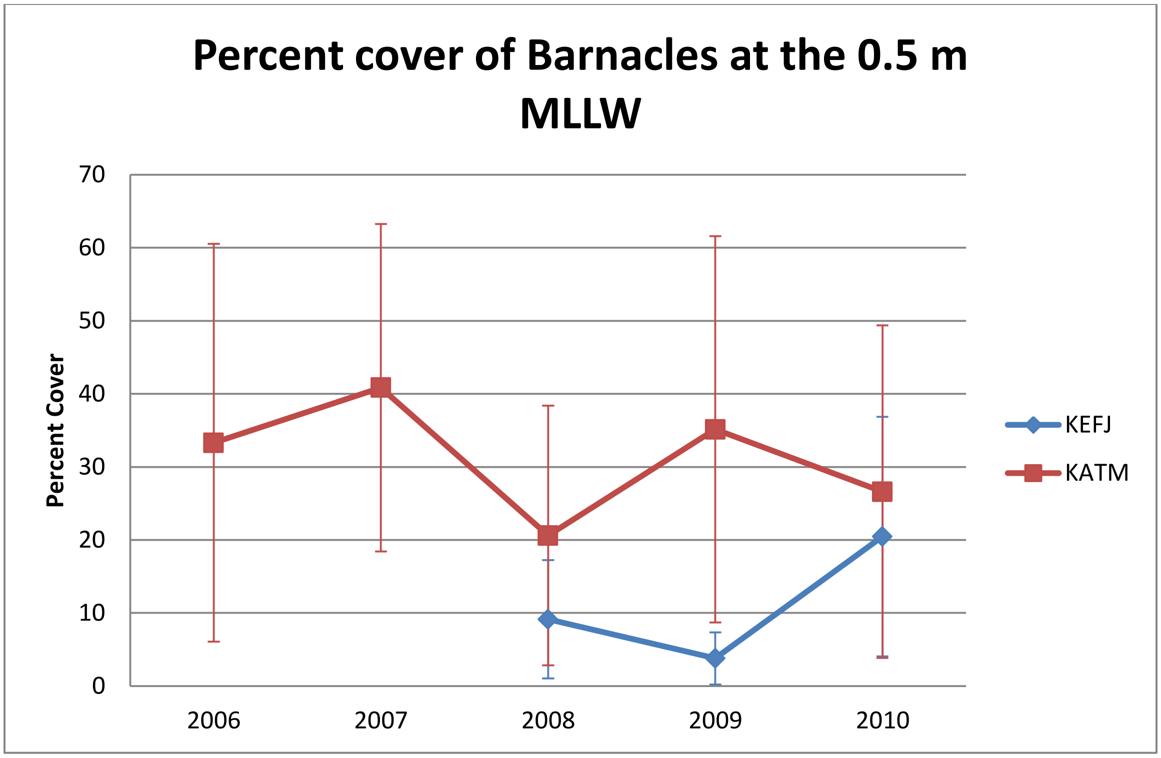


Figure 4. Percent cover of barnacles at the 0.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

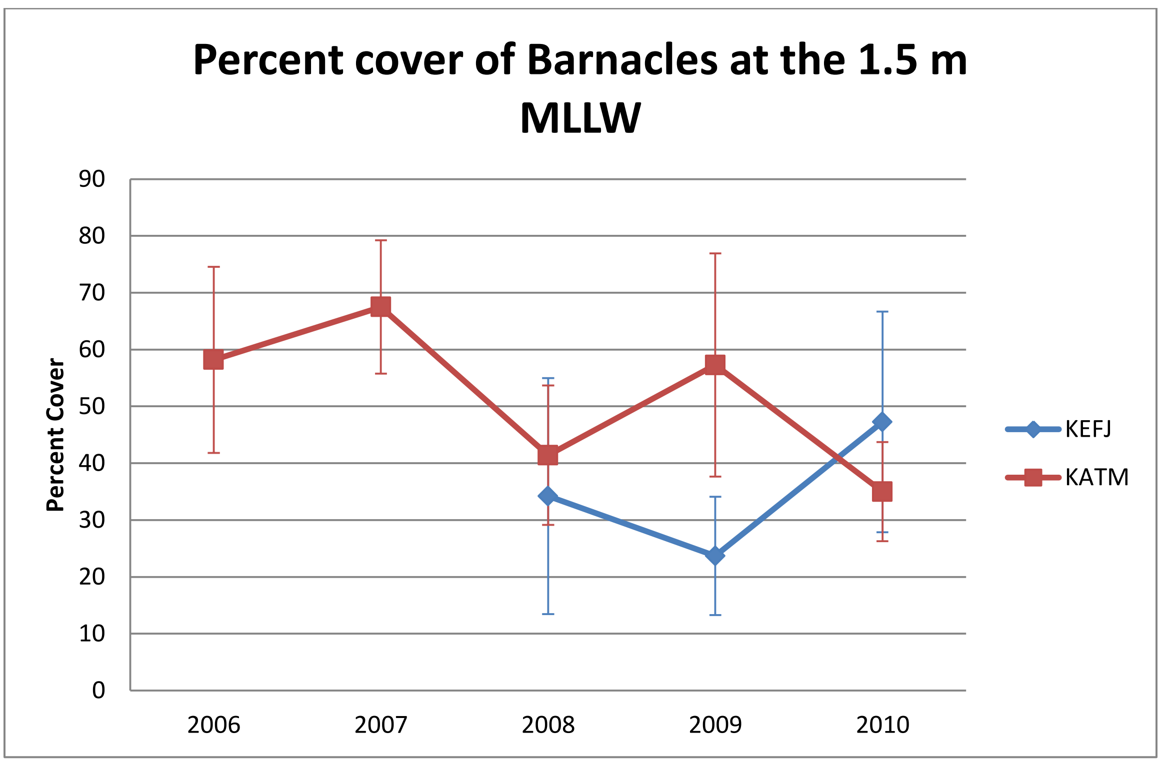


Figure 5. Percent cover of barnacles at the 1.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

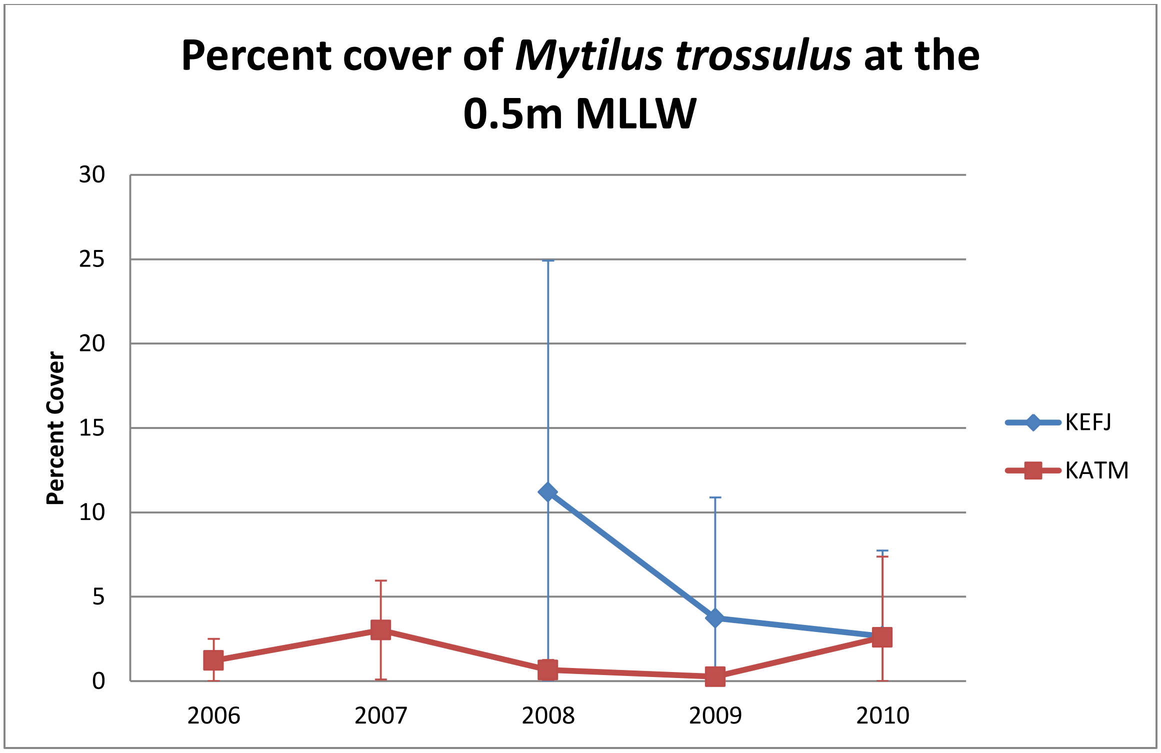


Figure 6. Percent cover of *Mytilus trossulus* at the 0.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

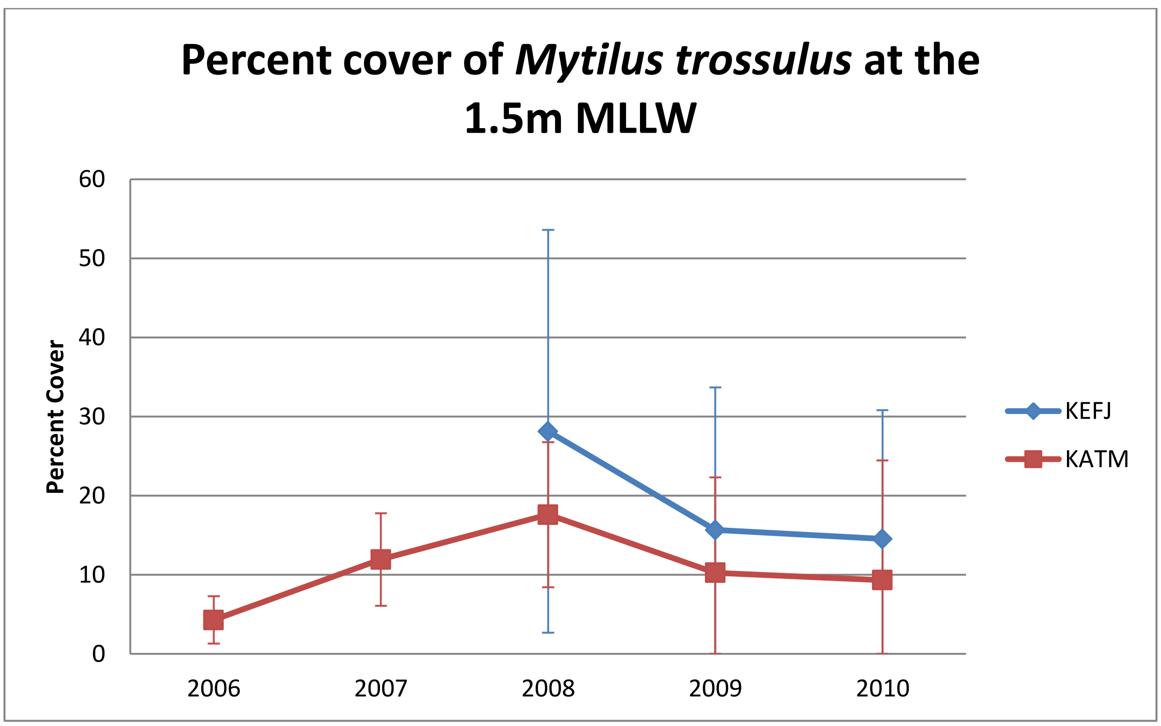


Figure 7. Percent cover of *Mytilus trossulus* at the 1.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

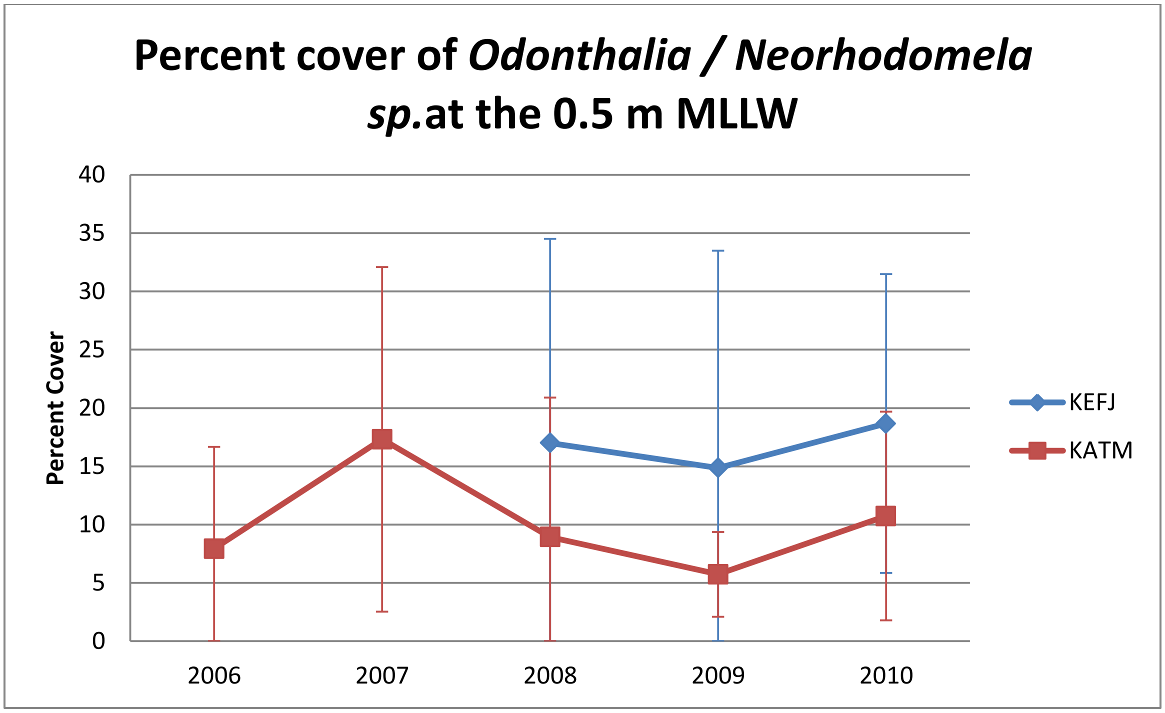


Figure 8. Percent cover of *Odonthalia / Neorhodomela* at the 0.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

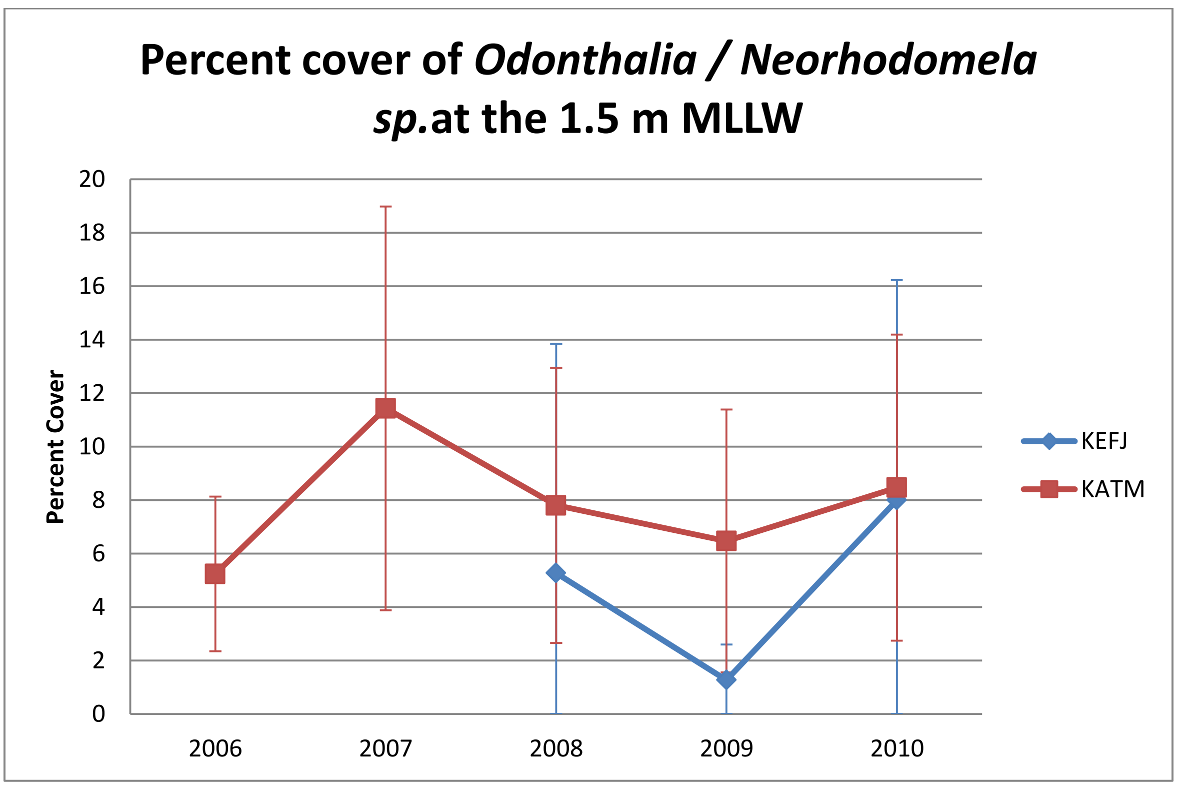


Figure 9. Percent cover of *Odonthalia / Neorhodomela* at the 1.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

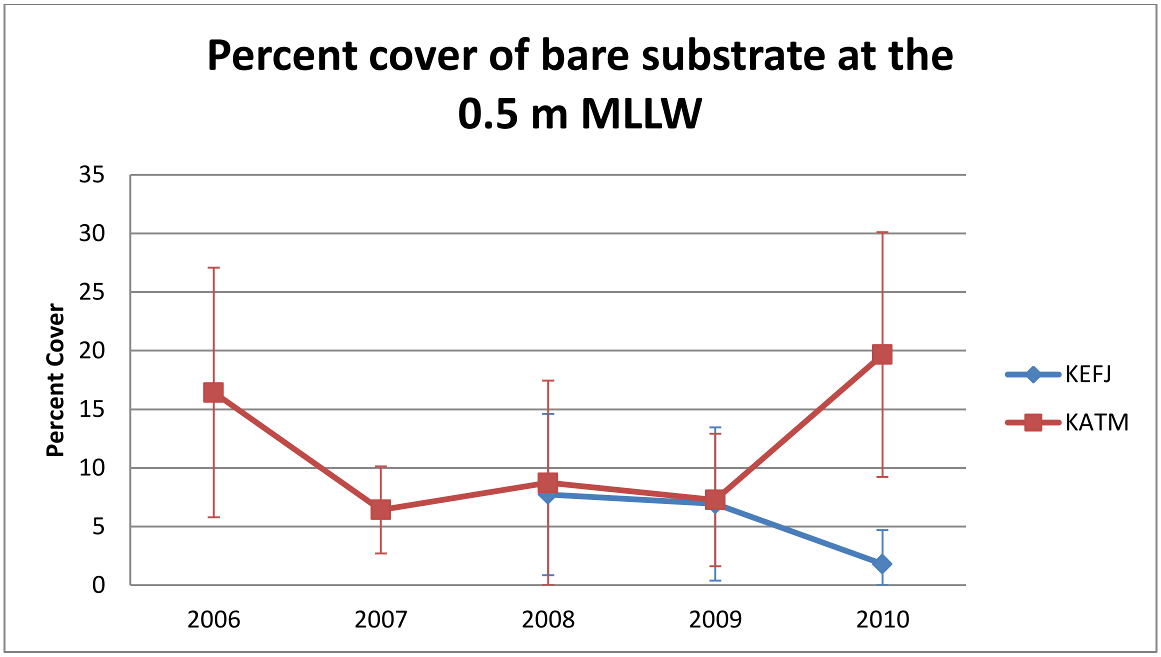


Figure 10. Percent cover of bare substrate at the 0.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

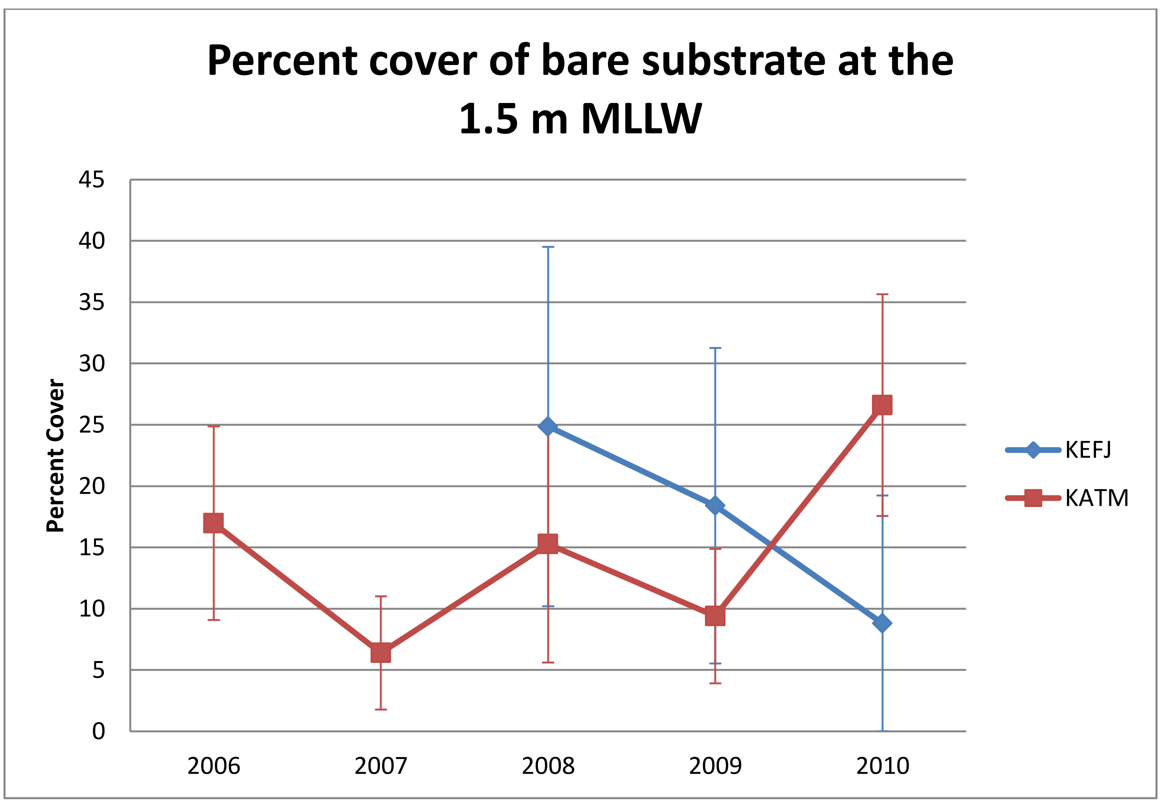


Figure 11. Percent cover of bare substrate at the 1.5 m MLLW in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

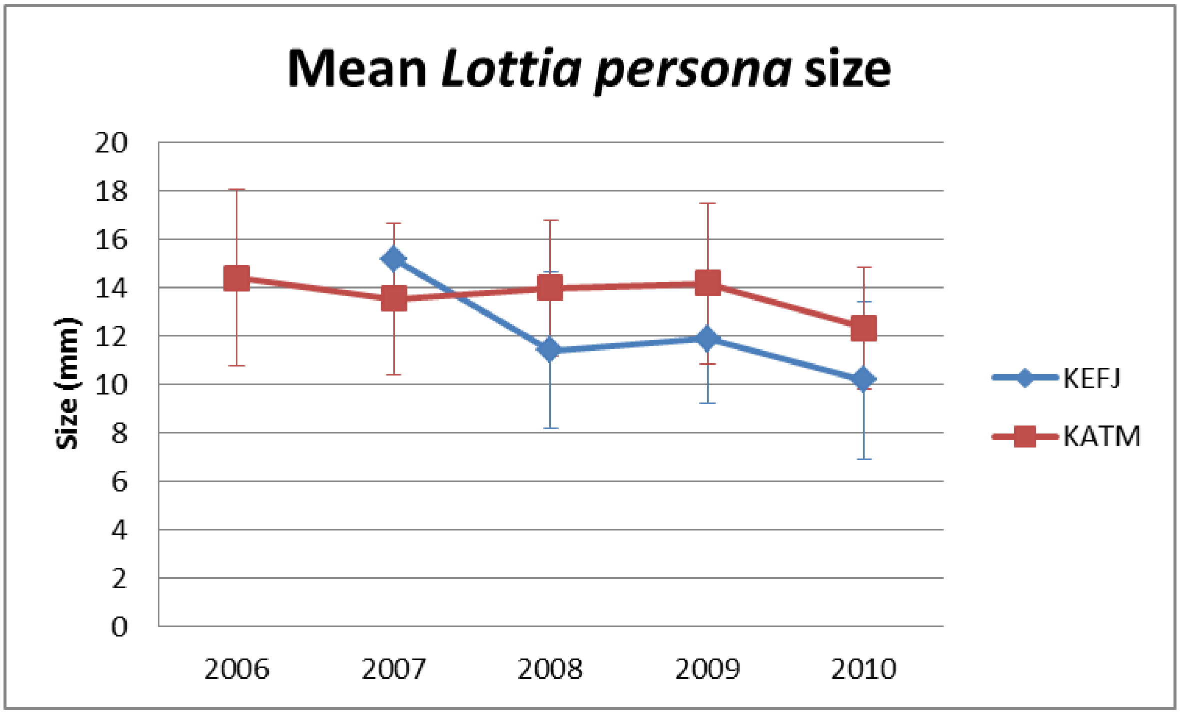


Figure 12. Mean size of *Lottia persona* in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI. There are no error bars for KEFJ in 2007 because only one site was sampled.

Discussion

The sampling described provided reasonable estimates of the abundance of intertidal invertebrates and algae (including intertidal kelps) at sites within each park. There were no clear trends in abundance over time, except for an increase in abundance of *Fucus* at KEFJ between 2008 and 2010. It is yet to be determined if this increase is a meaningful long-term trend. We anticipate that the methods employed will detect ecologically meaningful levels of change in the future. Existing data will allow the program to begin trend analysis for several metrics and will be used in simulations to estimate number of samples and sample frequency required to detect a specified trend or change with some level of confidence for selected metrics, specifically the rocky intertidal algae and invertebrate vital sign. The rocky intertidal invertebrate and algae vital sign has eight (8) metrics that that have several years of data to conduct simulations. The levels of change or trend have already been specified by the investigators (Dean and Bodkin 2011a). The Vital Signs Monitoring Plan for SWAN explicitly states the use of hierarchical models to estimate trends. The work proposed here is to assist the National Park Service in the modification of the protocol for its monitoring program.

Recommendations

Based on these results, we recommend continued estimation of percent cover by sessile invertebrates and algae using random point counts and continued estimation of sizes of limpets.

Mussel Bed Sampling

Introduction

Pacific blue mussels (*Mytilus trossulus*) are a dominant invertebrate in the intertidal zone and are critically important prey for a variety of organisms including sea otters, black oystercatchers, harlequin ducks, Barrows goldeneye, and several species of sea stars (O'Clair and Rice 1985, O'Clair and O'Clair 1988, VanBlaricom 1988, Andres and Flaxa 1995, Esler et al. 2002, Bodkin et al. 2002). Mussels are widely distributed in many intertidal habitats, but also form relatively monotypic stands of larger individuals that are termed mussel beds. The goal of mussel bed sampling is to assess changes in the size of beds and in the size of mussels within those beds over time. These data are primarily to be used as an indicator of mussel abundance as prey for various vertebrate predators (sea stars, sea ducks and sea otters). Specifically, the objectives are to estimate: 1) the density of mussels within these beds, 2) the density of large mussels within these beds, and 3) the size distribution of larger mussels within the beds (those generally consumed by black oystercatchers, sea ducks and sea otters). Sampling is conducted in sheltered rocky habitats within KATM and KEFJ. We define mussel beds as sites with relatively high densities of Pacific blue mussels. Specifically, mussel beds are defined as areas with greater than approximately 10% cover by mussels within contiguous 0.25 m^2 quadrats over areas of 100 m^2 or greater. Metrics used to evaluate change over time will include the area of individual mussel beds (in m^2), average density of large mussels (greater than 20 mm in length), and the mean size of mussels > 20 mm.

Methods

Sampling sites are defined as 50 m of coastline with contiguous mussel beds. These sites were selected following intensive searches in 2008 for the presence of mussel beds adjacent to the randomly selected rocky intertidal sites (see intertidal invertebrates and algae section). The closest mussel bed to the randomly selected rocky intertidal site was selected for sampling.

A transect 50 m in length was established through the mid-point of the bed, relative to tidal elevation, and at the left end of the bed, as observed from the water. A permanent bolt was placed at this location and at 5 m intervals along the 50 m length of the horizontal transect to establish the site for future sampling. Ten vertical transects were then established at systematic intervals based on a random start point along the horizontal transect length, and the distance from the upper most margin of the bed to the lower margin (or the 0 m tidal elevation) were measured.

Estimates of mussel density are made within quadrats that are randomly located along each vertical transect. Quadrat dimensions are dependent on the density of mussels $\geq$ 20 mm within 1 m of the predetermined random point along the vertical transect, and determined at the time of sampling. The quadrat size can range from .0025 m^2 to 1.00 m^2 (5 cm to 100 cm on a side) with the size dependent on obtaining a collection of at least 20 mussels $\geq$ 20 mm in length. This results in approximately 200 mussels to estimate size distributions. All mussels $\geq$ 20 mm are collected from within the quadrat and later counted and measured, and densities of large mussels are calculated. Densities of all mussels (of a size that is visually detectable, approximately 5 mm and greater) are estimated from a 2.54 cm radius (20.27 cm^2) core located at the same random number that defined the vertical quadrat, but on the opposite side of the tape from the origin of the quadrat.

Results

In 2010 we estimated the abundance and size of mussels at five mussel bed sites in KATM and at five sites in KEFJ for the third year in a row. Results for all three years at each park are represented here. In general, mussel density is greater in KEFJ than in KATM for all mussels as well as mussels ≥ 20 mm (Figures 13 and 14). Mussel sizes are greater in KATM than KEFJ, but not significantly (Figure 15). The proportion of mussels ≥ 20 mm has decreased in KATM, but increased in KEFJ (Figure 16). These results, however, are not significant.

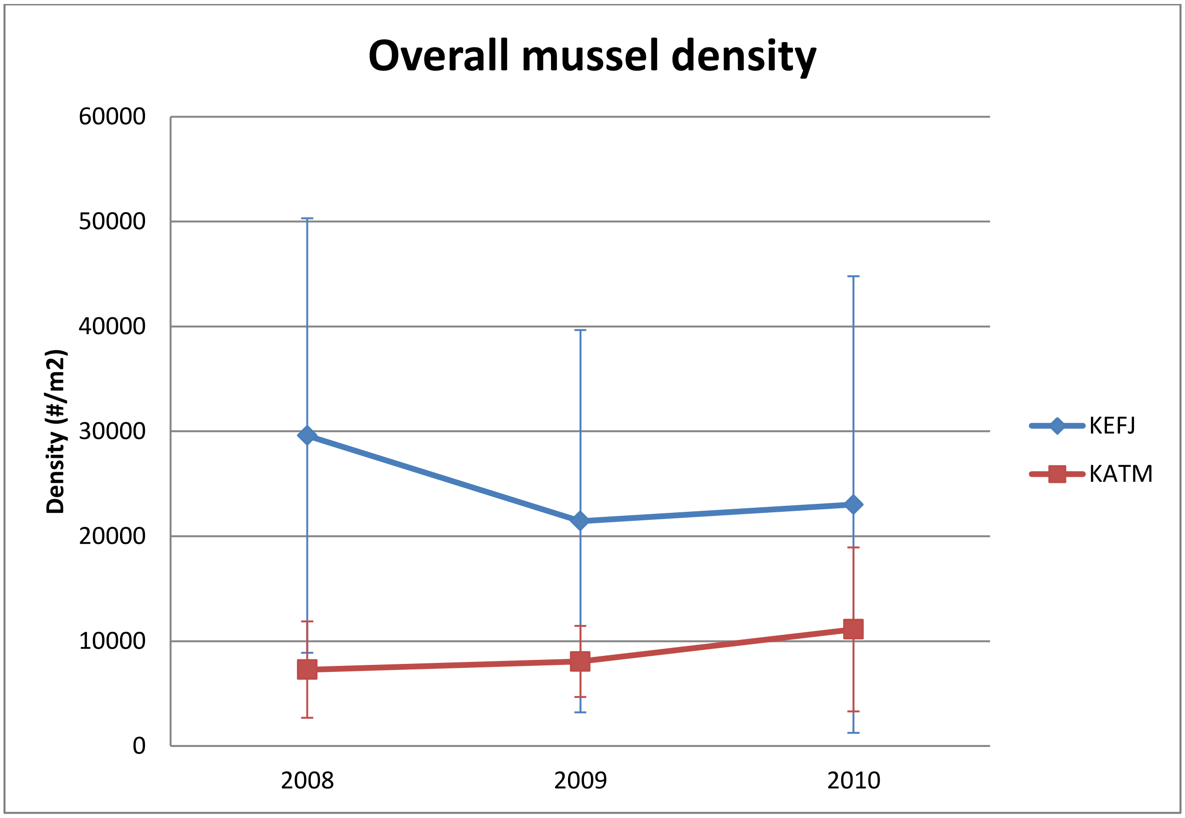


Figure 13. Overall mussel density (#/m^2) in KATM and KEFJ, 2008-2010. Error bars indicate 90% CI.

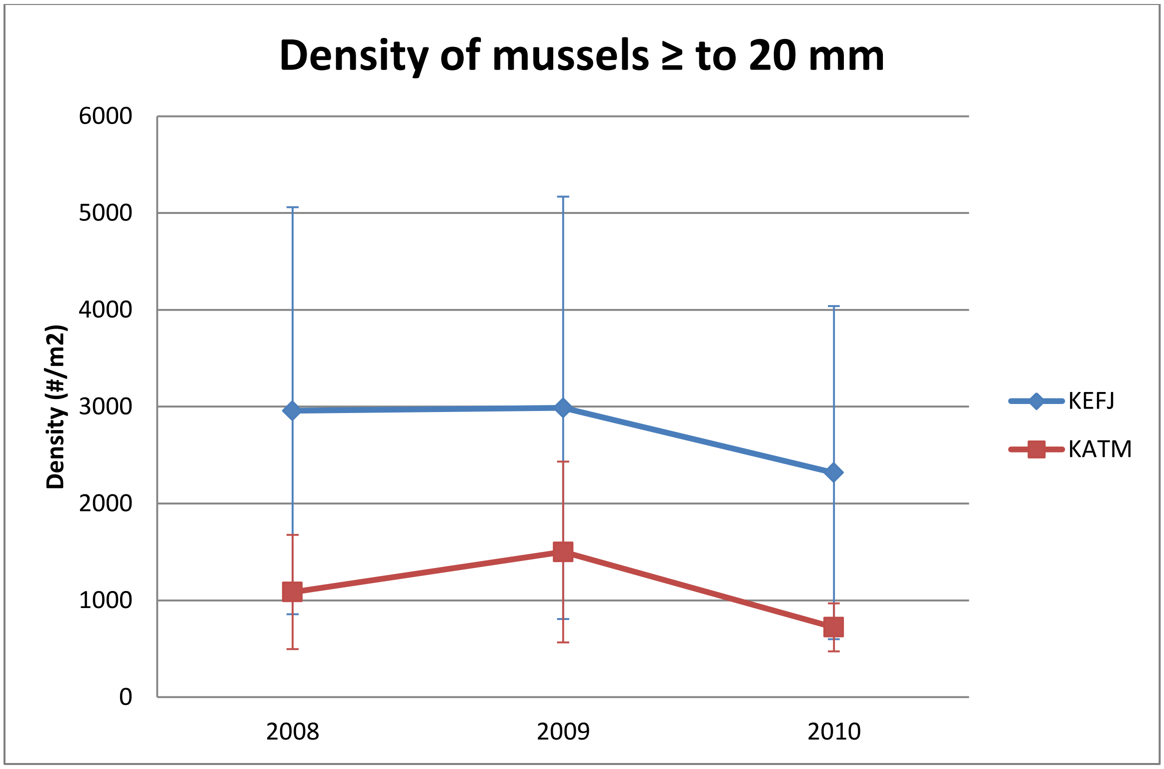


Figure 14. Density (#/m^2) of mussels ≥ 20 mm in KATM and KEFJ, 2008-2010. Error bars indicate 95% CI.

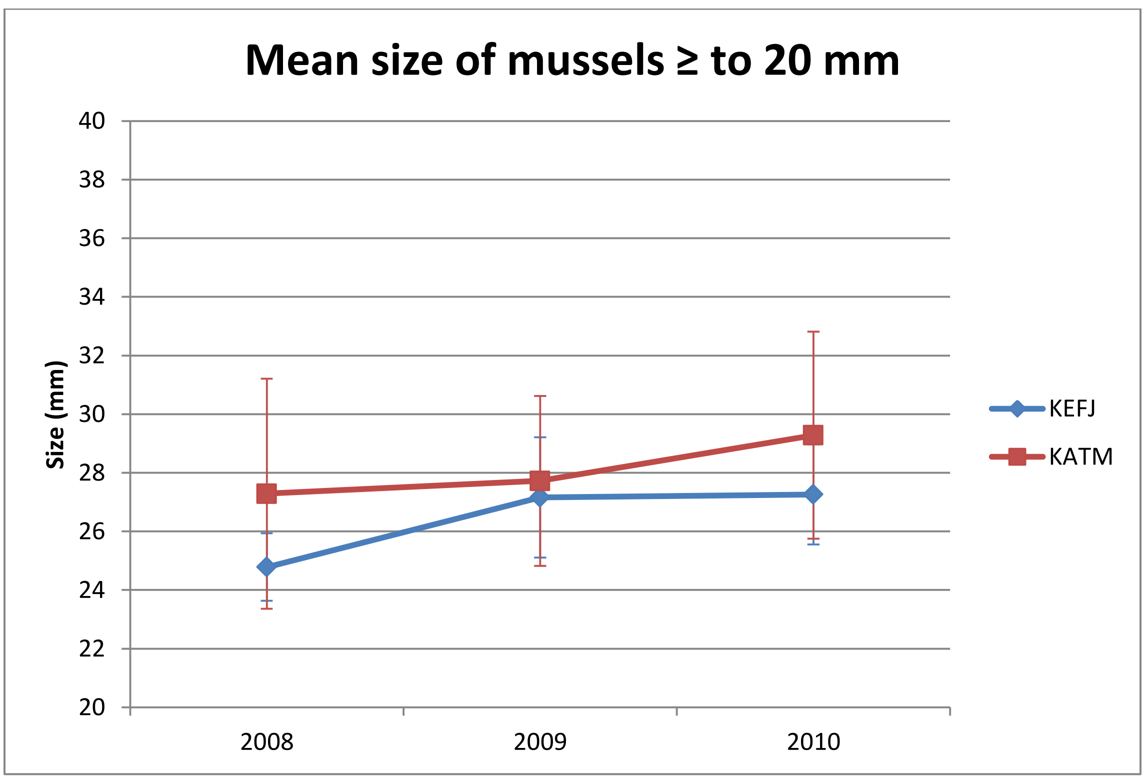


Figure 15. Mean size of mussels ≥ 20 mm in KATM and KEFJ, 2008-2010. Error bars indicate 95% CI.

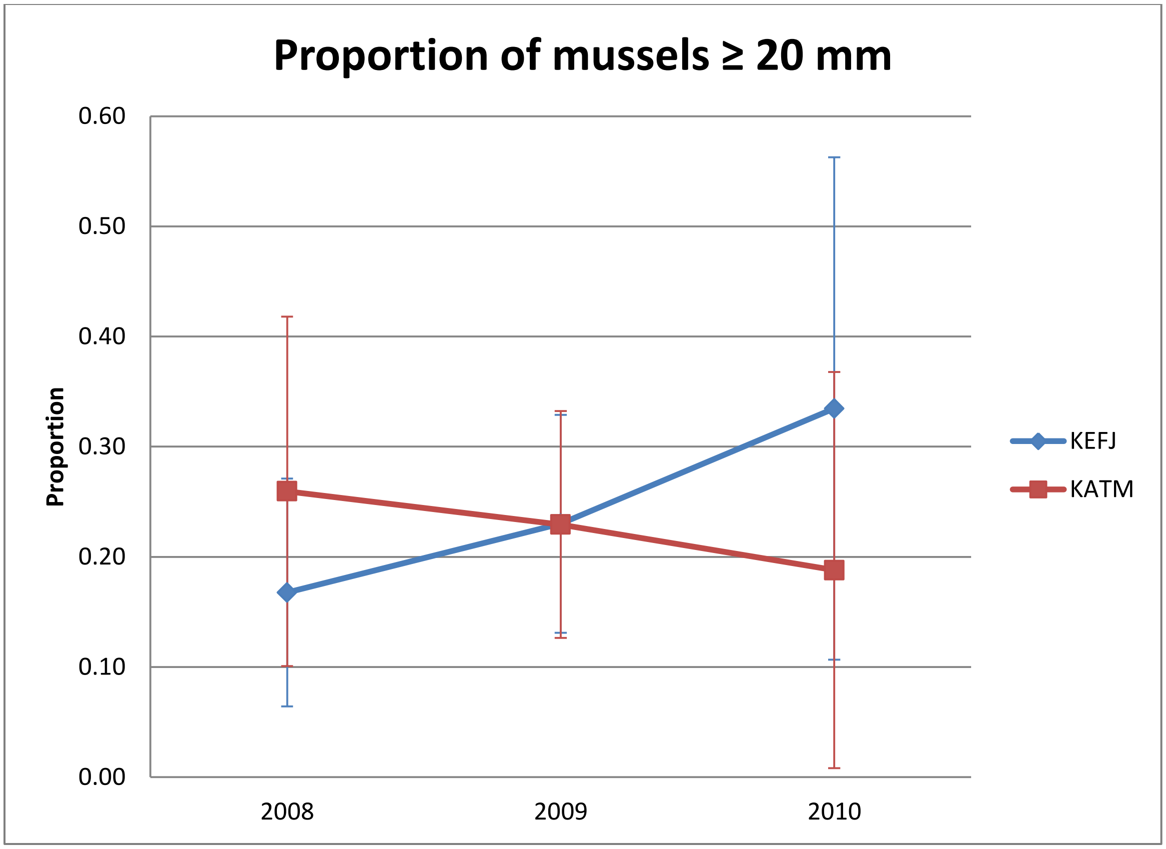


Figure 16. Proportion of mussels ≥ 20 mm in KATM and KEFJ, 2008-2010. Error bars indicate 90% CI.

Discussion

Using the methods briefly described above, we were able to estimate densities of mussels, the size distribution and density of mussels ≥ 20 mm, and the proportion of mussels ≥ 20 mm. Mussel densities varied greatly among parks, both in terms of all mussels and those ≥ 20 mm. Mean sizes of mussels ≥ 20 mm were relatively uniform among all sites, indicated by the smaller error bars. The high uniformity in mean sizes and low variance among sites, suggest perhaps a common mechanism structuring the sizes of mussels in the parks. While evaluating variance estimates of mussel densities and sizes for sensitivity to detect change will require additional years of data, the relatively low variation in mean sizes of large mussels across sites continues to suggest that mussel size may provide a statistically powerful metric to detect change over time.

Recommendations

Our third year of descriptive analysis indicates that sizes of mussels may provide a metric sensitive to change both among and within sites. We recommend the continuation of annual mussel bed sampling. Similar to the algae analysis discussed in the previous section, existing mussel bed data will allow the program to begin trend analysis for several metrics and will be used in simulations to estimate number of samples and sample frequency required to detect a specified trend or change with some level of confidence for selected metrics, specifically the rocky intertidal algae and invertebrate vital sign. The levels of change or trend have already been specified by the investigators (Dean and Bodkin 2011a). The Vital Signs Monitoring Plan for SWAN explicitly states the use of hierarchical models to estimate trends. The work proposed here is to assist the National Park Service in the modification of the protocol for its monitoring program.

Eelgrass Bed Sampling

Introduction

Eelgrass (*Zostera marina*) is the dominant seagrass in protected waters of the Gulf of Alaska and is broadly distributed in sheltered embayments, especially in habitats dominated by soft sediments where they often form “beds” or relatively monotypic stands that can cover much of the shallow (0 to 5 m depth) subtidal zone (McRoy 1968, 1970). Eelgrass is an important "living habitat" that serves as a nutrient filter, provides shelter for fish and a variety of invertebrates, and provides physical substrate for invertebrates and algae (Thayer and Phillips 1977, Jewett et al. 1999, Dean et al. 2000, Bostrom et al. 2006). Eelgrass is a major primary producer in the marine nearshore (McConnaughey and McRoy1979) and because it is located in shallow water, is susceptible to oil spills and other human disturbances (Short and Wiley-Eschevaria 1996, Dean et al. 1998, Duarte 2002, Larkum et al. 2006, Short et al. 2006). Eelgrass is especially susceptible to dredging, anchor scars, and events that reduce light penetration into the water column such as runoff (increased turbidity) or nutrient addition (Walker et al. 1989, Oleson 1996, Hauxwell et al. 2003, Neckles et al. 2005, Terrados et al. 2006).

The purpose of this sampling is to assess changes in the extent of eelgrass over time. In this report, we examine results from sampling eelgrass cover in KATM and KEFJ in 2010. The sampling is designed to examine smaller spatial (within beds of approximately 1 km^2) over temporal scales of several years.

Methods

We sampled the percent cover of eelgrass at four sites in KATM and four sites in KEFJ in 2010. Future sampling will consist of annual visits to 5 sites in each park. The same designated area will be sampled at each site in each year. All sampling will be conducted in early summer when eelgrass beds generally have reached there seasonal maximum in extent and density of plants.

All beds sampled were in sheltered bays and were at eelgrass beds in closest proximity to sites selected for sampling of invertebrates on sand/gravel beaches that were chosen using a GRTS procedure (Stevens and Olsen, 2004) that provided a spatially balanced yet random selection of sites. At each site we sampled eelgrass within a prescribed area along a shoreline of approximately 200 m in length. The width of each bed examined depended on the depth contour at each site, but was generally on the order of 50 to 100 m. The areas sampled were bounded by an approximately 200 m segment of shoreline over which eelgrass was observed and extended offshore to a distance approximately 15 m beyond the last observed eelgrass. The percent cover of eelgrass within this area was estimated by determining the presence or absence of eelgrass at approximately evenly spaced intervals along a series of transects running perpendicular to shore that were spaced approximately 20 m apart. Presence or absence at each observation point was determined using an underwater video camera lowered from a small inflatable boat and a single-beam sonar.

These surveys will allow us to detect changes in average extent of eelgrass over time. While we do not know the types of changes that might occur, these might include local reduction in cover due to increased boating activity and associated anchor scars, a lowering of the upper depth limitation due to a decline in water clarity, or larger scale die offs due to diseases or contaminants.

Results

The percent of observations with eelgrass present ranged from 25 to 73%. The highest percent covers were observed at Amalik Bay in KATM and Harris Bay in KEFJ.

Table 1. Percent of observations with eelgrass at sites in KATM and KEFJ in 2010. Means and 90% confidence intervals (mean plus or minus CI) are given. Dots indicate 'no data'.

Park Name	Site Name	Site Number	Percent of observations with eelgrass present
KATM	Kukak	AP-B10-RI1	56
	Kaflia	AP-B10-RI2	50
	Amalik	AP-B10-RI4	60
	Takli	AP-B10-RI5	38
	Mean		51
	CI		8
KEFJ	Aialik	KP-B5-RI1	25
	McCarty	KP-B5-RI2	32
	Nuka Bay	KP-B5-RI3	67
	Harris	KP-B5-RI5	73
	Mean		49
	CI		20

Discussion

Using the methods briefly described above, we were able to estimate percent cover by eelgrass in prescribed areas. Determination of our ability to detect change in eelgrass cover over time will require additional years of sampling.

Recommendations

Based on replicate sampling completed in 2008, our analysis indicated that the method produces relatively precise estimates of the relative abundance of eelgrass. We recommend the continuation of annual eelgrass bed sampling.

Marine Bird Surveys

Introduction

Marine birds and mammals are important constituents of marine ecosystems and are sensitive to variation in marine conditions. Our focus on nearshore marine bird monitoring will be on species that are relatively abundant and trophically linked to the nearshore food web where the kelps and seagrasses contribute substantially to primary productivity and benthic invertebrates such as clams, mussels and snails, transmit that energy to higher level trophic level fishes, birds and mammals. Species of focus in the nearshore food web include black oystercatchers, cormorants, glaucous-winged gulls, black-legged kittiwakes, goldeneyes (winter density and distribution), harlequin ducks, pigeon guillemots, mergansers and scoters. Because other birds and mammals will be encountered in the course of monitoring nearshore species, observations of all marine birds and mammals are recorded.

The sea ducks and black oystercatcher were selected for focus because of their reliance on habitats and prey associated with nearshore marine communities. These species are top level consumers of nearshore invertebrates such as mussels, clams, snails, and limpets that are being monitored under the algal and intertidal invertebrate SOP. These species play an important role as consumers of marine invertebrates (Draulans 1982, Marsh 1986a and b, Meire 1993, Lindberg et al. 1998, Hamilton and Nudds 2003, Lewis et al. 2007). Therefore, understanding changes in their abundance over time is an important metric for nearshore monitoring. Abundance estimates will be enhanced by the nearshore invertebrate monitoring SOP, which focuses on sampling their prey populations. Moreover, monitoring trends in abundance of the various guilds of other marine birds (e.g. pigeon guillemots, black-legged kittiwakes, and cormorants) that utilize other food sources may improve the ability to discriminate among potential causes of change in seabird populations and the nearshore ecosystem. For example, concurrent changes in sea ducks, which forage on nearshore invertebrates, and the pigeon guillemots that forage on small fish, may suggest a common cause of change, one that may be independent of food. Such an approach may provide insights related to competing hypotheses relative to cause of change within or among populations (Petersen et al. 2003). In addition many of these species, including the harlequin duck, Barrow’s goldeneye, and black oystercatcher were impacted by the Exxon Valdez oil spill, and exhibited protracted recovery periods as a consequence of lingering oil in nearshore habitats in Prince William Sound (Andres 1999, Trust et al. 2000, Esler et al. 2000a and b, Esler et al. 2002). Long-term monitoring of these species at different locations will likely provide increased confidence in assessment of the status of these populations relative to restoration and recovery from the 1989 spill. Additionally, existing data collected using comparable methods are available from other nearshore habitats in the Gulf of Alaska for periods up to 20 years (Irons et al.1988, Irons et al. 2000). Long-term monitoring of these species at different locations will likely provide increased confidence in assessment of the status of these populations relative to restoration and recovery from the 1989 spill.

Methods

Standardized surveys of marine birds were conducted in KATM (2006-2010) and KEFJ (2007-2010) between late June and early July. Surveys are conducted from small vessels (5-8 m length) traveling at speeds of 8-12 knots along selected sections of coastline that represent independent transects. The transect width is 200 m and the boat represents the midpoint. Transects are surveyed by a team of three. The boat operator generally surveys the 100 m offshore area of the transect, while a second observer surveys the 100 m nearshore area. The third team member enters the observations into a laptop running dLOG, specifically designed for this type of surveying, and assists with observations. All marine birds and mammals within the 200 m transect width are identified and counted. All transects considered in this analysis are run 100 m offshore and parallel to the shoreline. Detailed descriptions of methods and procedures can be found in the Marine Bird and Mammal Survey SOP (Bodkin 2011a).

The survey design consists of a series of transects along shorelines such that a minimum of 20% of the shoreline is surveyed. Transects are systematically selected beginning at a random starting point from the pool of contiguous 2.5-5 km transects that are adjacent to the mainland or islands, plus the lengths of transects that were associated with islands or groups of islands with less than 5 km of shoreline.

Each species is identified as important to nearshore food webs and as important indicators of change (Dean and Bodkin 2011a). Several species were grouped into higher order taxa (e.g., cormorants, mergansers and scoters) because identification to species within these groups was not always possible. Cormorant species included pelagic, red-faced, and double crested cormorants. Merganser species include common merganser and red-breasted mergansers. Scoters included surf, black, and white-winged scoters.

Results

Summer surveys were conducted in both KATM and KEFJ in 2010. Only focal species densities and standard errors observed on nearshore transects are reported here (Figures 17-24).

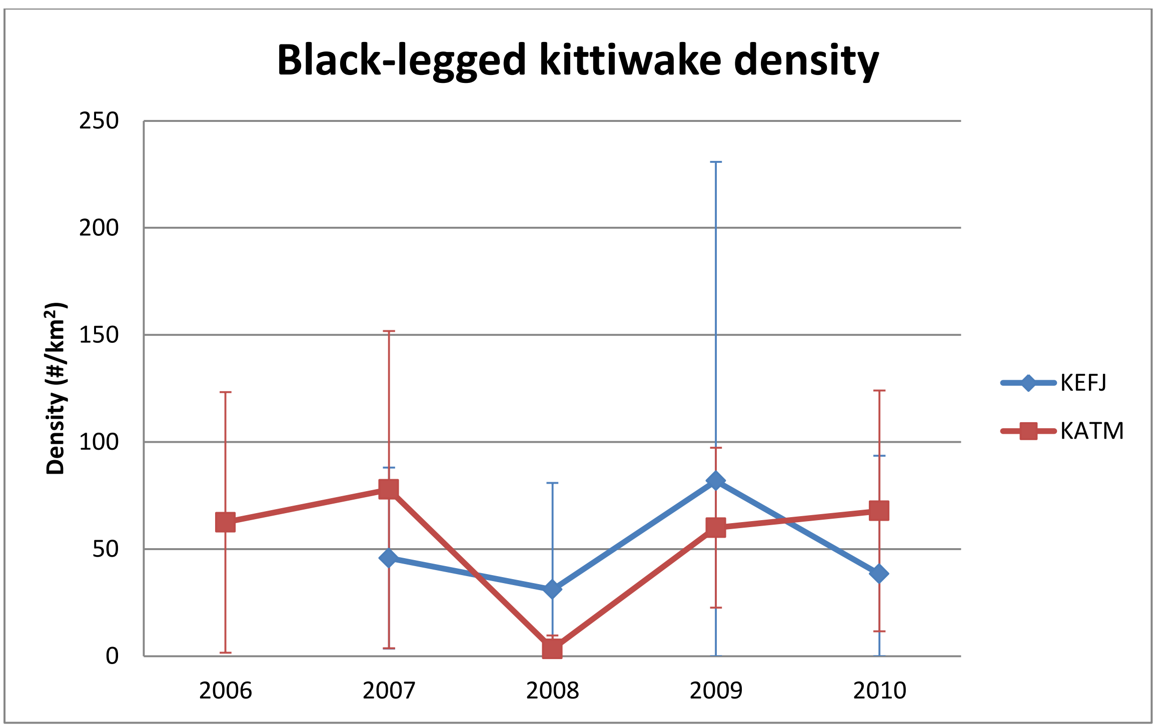


Figure 17. Density of black-legged kittiwake in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

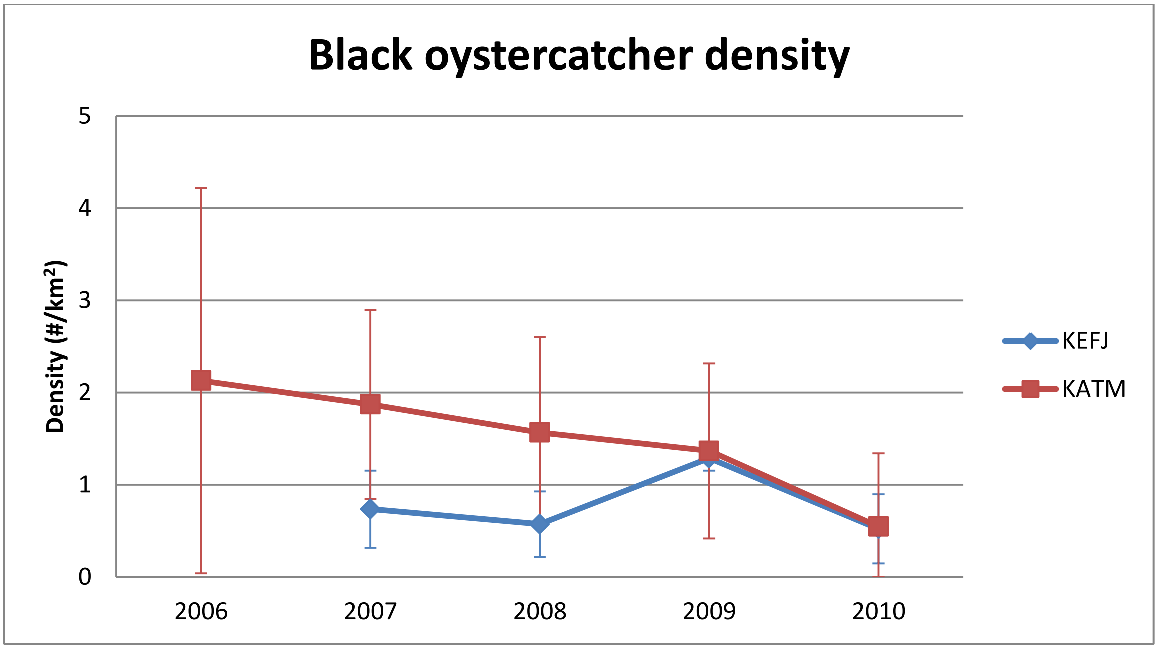


Figure 18. Density of black oystercatcher in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

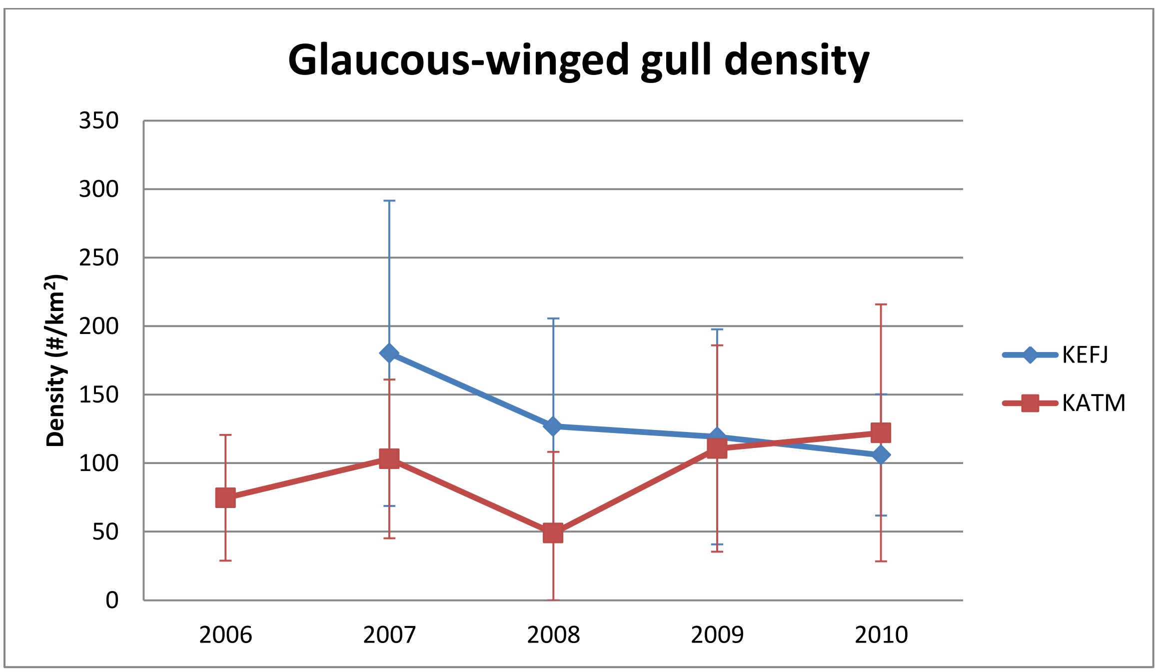


Figure 19. Density of glaucous-winged gull in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

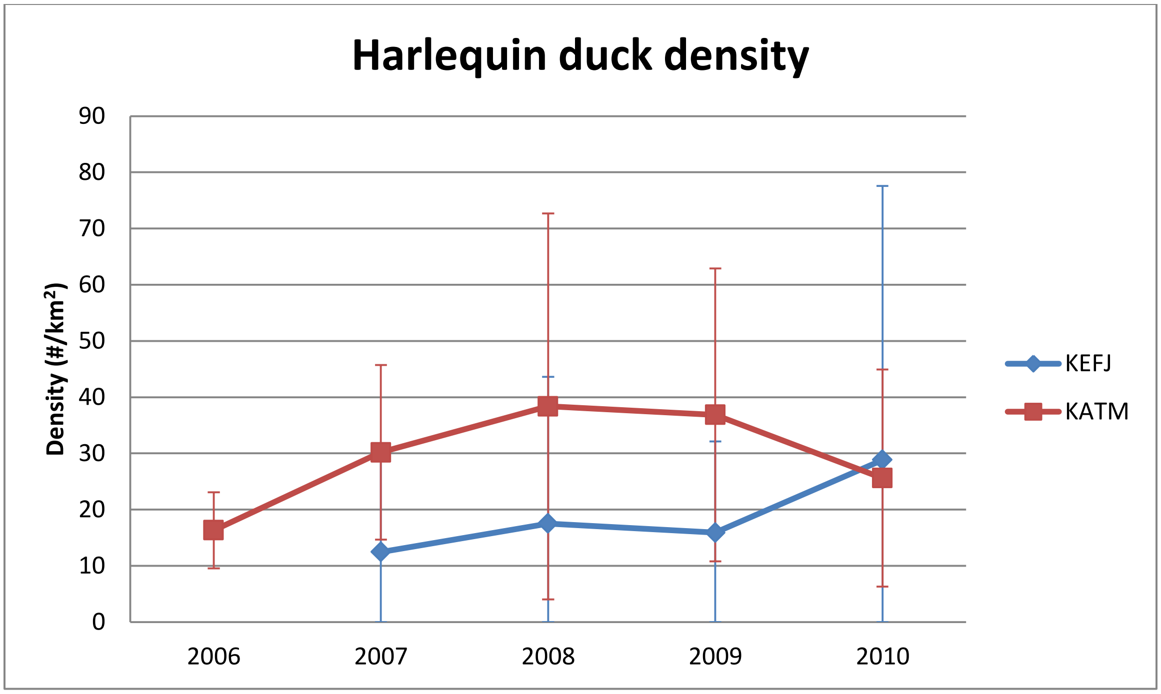


Figure 20. Density of Harlequin duck in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

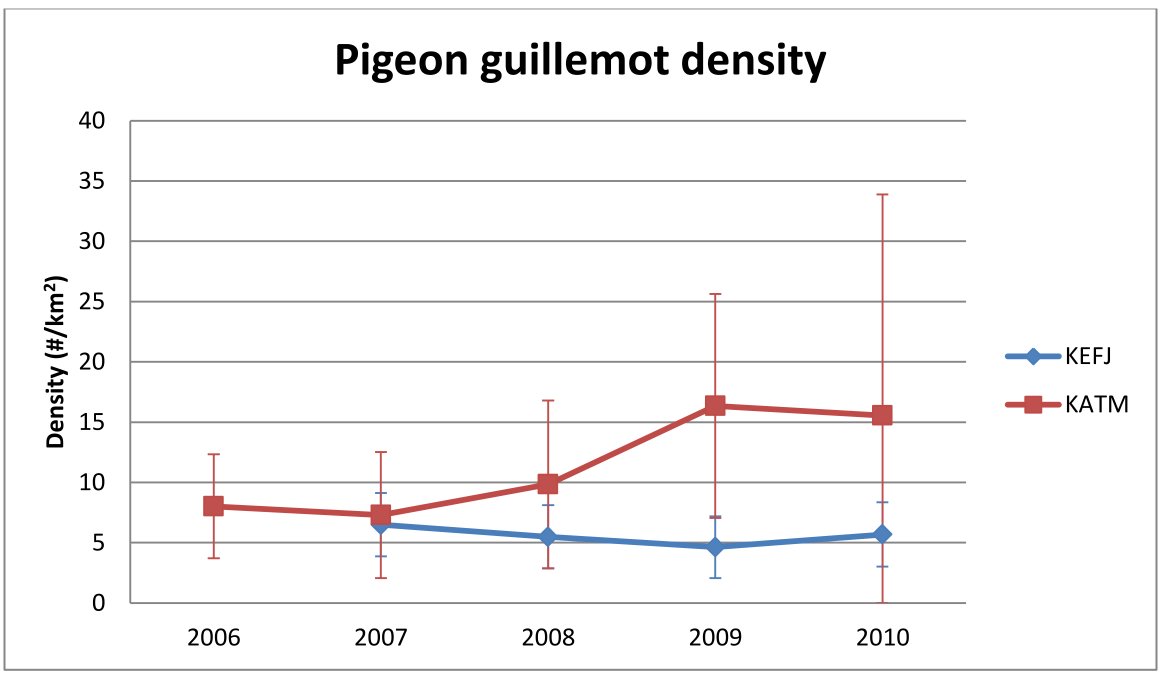


Figure 21. Density of pigeon guillemot in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

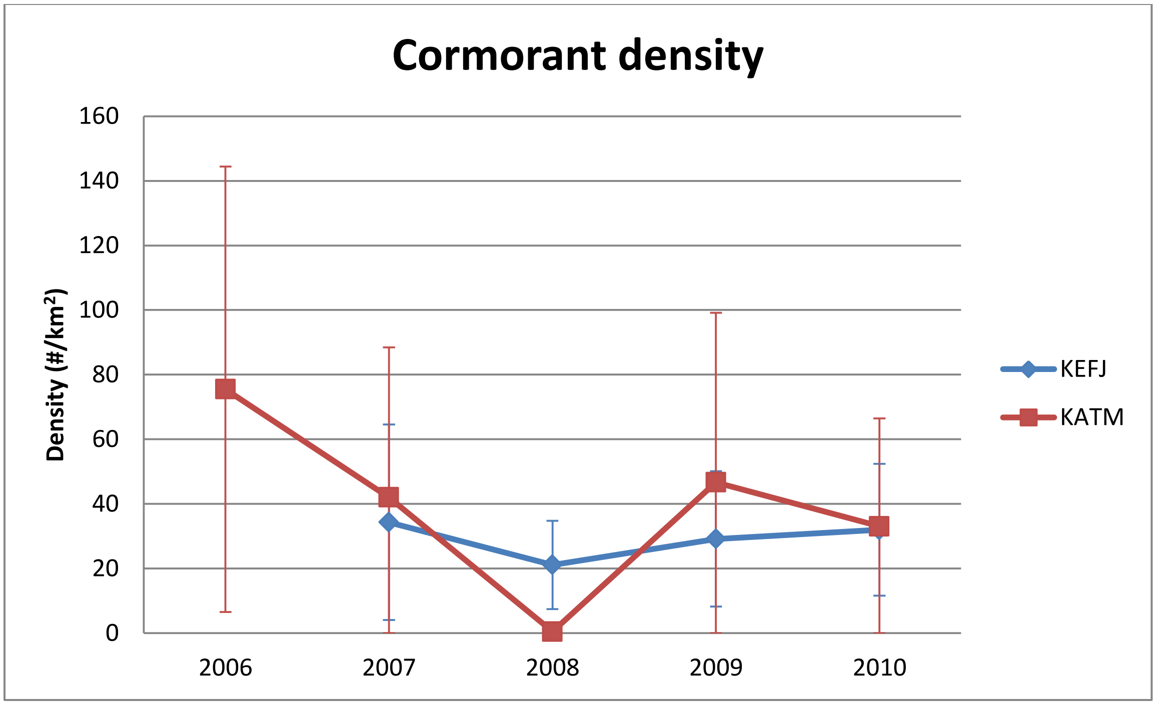


Figure 22. Density of cormorants in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

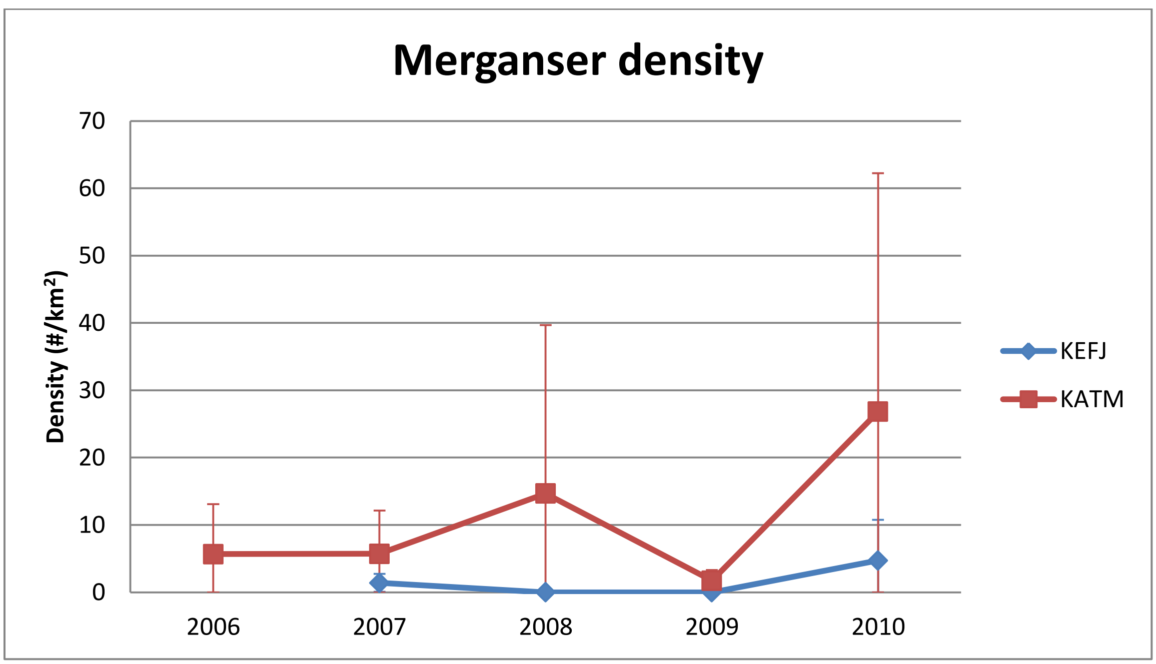


Figure 23. Density of mergansers in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

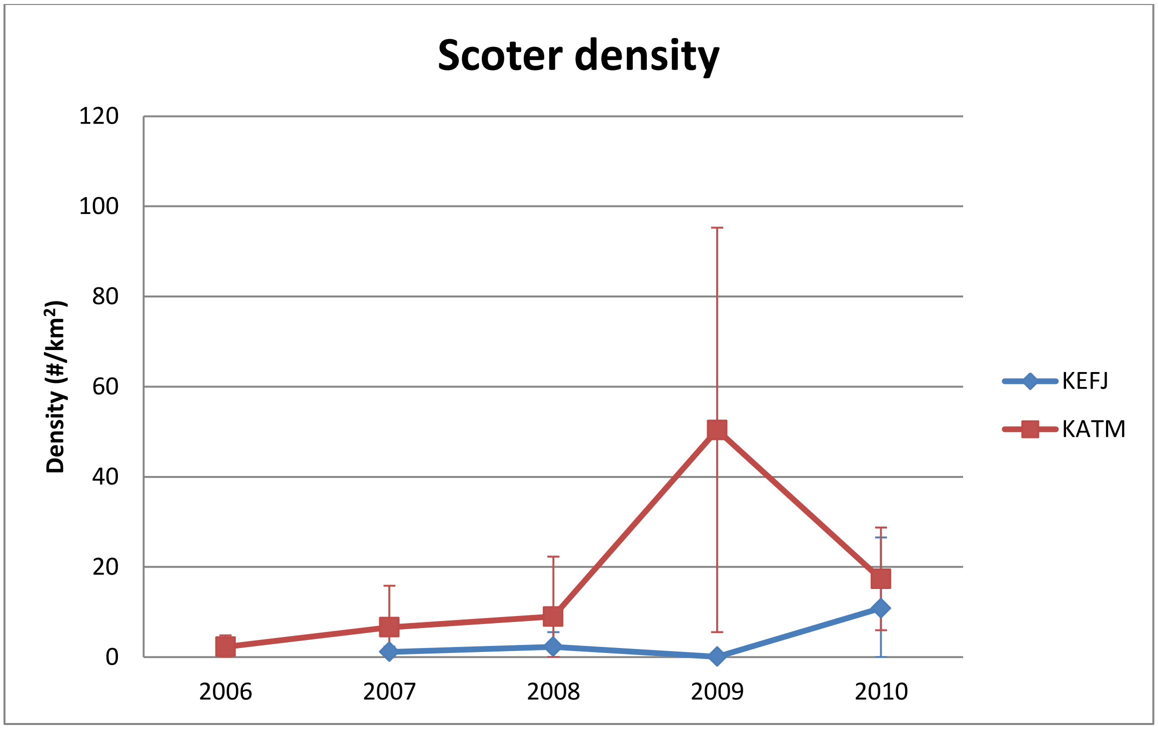


Figure 24. Density of scoters in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

Discussion

KATM and KEFJ continue to be sampled annually during the summer. These shoreline skiff surveys provide baseline information on species composition, distribution and density for summer populations of marine bird and mammal fauna that occur in the nearshore waters of KATM and KEFJ. Because components of the marine bird and mammal fauna may change seasonally, inference of species composition, distribution, and densities to other seasons cannot be made. In particular it is likely that some sea duck species that were rare or absent in the summer may be more common as over wintering residents (e.g. goldeneye, scoters, and long tailed ducks). Sustainability of long-term monitoring programs requires the optimization of sampling intensity and efforts to minimize costs while concurrently having sufficient power to detect a trend. While there has been critical thought in the past regarding these questions, current available analytical methods now allow for the use of existing data in simulations, using a Bayesian framework, to estimate number of samples and sample frequency required to detect a specified trend as well as examine effects contributing to variation, such as imperfect detection. An optimization exercise using existing data will occur in 2011-2012.

Recommendations

We recommend that survey effort continue until further analysis can be completed. These datasets will be examined to determine levels of change we can reasonably expect to detect based on this sampling method. We will also explore the possibility of re-allocating sampling efforts to specific habitat types or incorporate replicate sampling to enhance our ability to detect trends for species of interest.

Black Oystercatcher Sampling

Introduction

The black oystercatcher is a common and conspicuous member of the rocky and gravel intertidal marine communities of eastern Pacific shorelines and is completely dependent on nearshore marine habitats for all critical life history components including foraging, breeding, chick-rearing, and resting (Andres and Falxa 1995). During the late spring and summer breeding season pairs establish and defend both nest and forage areas, and these territories and nest sites can persist over many years (Groves 1984, Hazlitt and Butler 2001) with individual life expectancy exceeding 15 years (Andres and Falxa 1995). The diet consists primarily of mussels (*Mytilus* sp.) and a variety of limpets (*Lottia, Acmea*, and *Colisella* sp.) (Andres and Falxa 1995), which are ecologically and culturally important constituents of the intertidal community. The species is considered a Management Indicator Species by the Chugach National Forest and a species of concern nationally (Brown et al. 2001) and regionally (Alaska Shorebird Working Group 2000), and is widely recognized as a species representative of nearshore habitats. Because of their complete reliance on intertidal habitats, their reproductive biology, and foraging ecology, black oystercatchers are particularly amenable to long-term monitoring (Lentfer and Maier 1995, Andres 1998).

As a “keystone” species (Power et al. 1996), the black oystercatcher has a large influence on the structure of intertidal communities that is disproportionate to its abundance. The black oystercatcher receives its recognition as a keystone species through a three-trophic-level cascade initiated by the oystercatcher as a top level consumer in the nearshore (Marsh 1986a and b, Hahn and Denny 1989, Falxa 1992) whose diet consists largely of gastropod (limpets) and bivalve (mussels) mollusks that are ecologically important in the intertidal community. As a consequence of oystercatcher foraging, large numbers of herbivorous limpets can be removed (Frank 1982, Lindberg et al. 1987), resulting in shifts in limpet species composition and reduced size distribution (Marsh 1986a, Lindberg et al. 1987). As a consequence of reduced limpet densities and the diminished grazing intensity that results, algal populations respond through increased production and survival, resulting in enhanced algal populations (Marsh 1986a, Meese 1990, Wootton 1992, Lindberg et al. 1998). Additionally, like other invertebrate, avian and mammalian predators in the nearshore, a large fraction of the oystercatcher’s diet consists of mussels, an important filter feeding bivalve (Knox 2000, Menge and Branch 2001). Because the oystercatcher brings limpets, mussels and other prey back to its nest to provision chicks (Webster 1941, Frank 1982, Hartwick 1976, Lindberg et al. 1987), collections of those shell remains at nests provides an opportunity to obtain an independent sample of the species composition and size distribution of common and important nearshore invertebrate prey species that are directly estimated under intertidal algal and invertebrate vital signs (Intertidal Invertebrates and Algae section of this report). The collection of black oystercatcher diet and prey data offers a unique perspective into processes structuring nearshore communities (Marsh 1986a and b, Lindberg et al. 1987), including the potential consequences of anticipated increases in human presence and disturbance (Lindberg et al. 1998). Further, contrasting relative abundances and size-class composition of invertebrates collected under two independent protocols should increase our understanding of the processes responsible for change in nearshore ecosystems.

At a global scale, intertidal communities have been impacted by human activities (Liddle 1975, Kingsford et al. 1991, Povery and Keough 1991, Keough et al. 1993, Menge and Branch 2001) and one of the primary capabilities and intents of the nearshore monitoring program is to provide early detection of change in nearshore communities and to separate human from natural causes of change. Because of the critical nature of intertidal habitats for both breeding and foraging, black oystercatchers are particularly sensitive indicators to disturbances in the nearshore (Lindberg et al. 1998). Specifically, black oystercatchers nest exclusively in the intertidal, where eggs are laid in exposed nests consisting of depressions in pebbles, sand, gravel, and shell materials. During the 26-32 d incubation phase of reproduction, eggs are susceptible to predation by other birds (primarily C*orvids*; Lentfer and Meier 1995) and mammals (Vermeer et al. 1992), as well as human disturbance and trampling. Similar disturbance effects occur during the chick rearing stage, which lasts approximately 38 d (Andres and Falxa 1995). Thus, for several months during May-August, typically when human presence in nearshore habitats in Alaska is highest, black oystercatchers are actively incubating or caring for young in a habitat that affords little protection from human induced disturbances. Chronic disturbance from human activities poses a significant threat to breeding black oystercatchers, either preventing nesting altogether, causing nest abandonment after eggs have been laid (Andres 1998), or through direct mortality of eggs or chicks. Monitoring of black oystercatcher density, breeding territory density and occupancy, and prey will provide a potentially powerful tool in identifying the magnitude and causes of inevitable change in Gulf of Alaska nearshore habitats and communities, particularly in response to the anticipated increased use and influence of those habitats by humans.

Methods

There are three components to the sampling related to black oystercatchers: estimation of breeding pair density and nest occupancy through oystercatcher-specific surveys; estimation of species composition and size distributions of prey returned to provision chicks; and estimation of density of breeding and non-breeding black oystercatchers observed during the marine bird and mammal surveys. Results regarding the black oystercatcher density estimates are given in the marine bird survey section of this report. Detailed survey methods for estimation of nest occupancy and diet can be found in the black oystercatcher breeding territory occupancy and chick diet SOP (Bodkin 20011b). The detailed methods used to obtain marine bird densities can be found in the marine bird SOP (Bodkin 2011a) and in Bodkin et al. (2007b and 2008).

Black oystercatcher breeding territory density, nest occupancy, and prey data were collected along five 20 km transects each centered on the randomly (GRTS) rocky intertidal algal and invertebrate sites at KATM since 2006 and KEFJ since 2007. Nest sites were located by surveying the shoreline in a small boat. All accessible nest sites were visited to determine the number of chicks and/or eggs present and all prey items (e.g. mussel or limpet shells) present at a nest site were collected. All prey were measured. Here, we present size data for most abundant prey species, Pacific blue mussels (*Mytilus trossulus*) and the limpets (*Lottia pelta*, *Lottia persona* and *Lottia scutum*).

Results

Density and Productivity

All five black oystercatcher GRTS transects were analyzed at the park level for nest density (nest/km) and productivity (chicks + eggs/nest) by year. The mean density of active black oystercatcher nest sites at KATM ranged from 0.05 to 0.11 per km of shoreline from 2006-2010 (Figure 25). The mean density of active black oystercatcher nest sites at KEFJ ranged from 0.05 to 0.10 per km of shoreline from 2007-2010 (Figure 25). The mean productivity (eggs + chicks / nest) ranged from 1.42 to 2.3 eggs + chicks / nest for KATM from 2006-2010 (Figure 26). The mean productivity (eggs + chicks / nest) ranged from 0.27 to 1.92 eggs + chicks / nest for KEFJ from 2007-2010 (Figure 26).

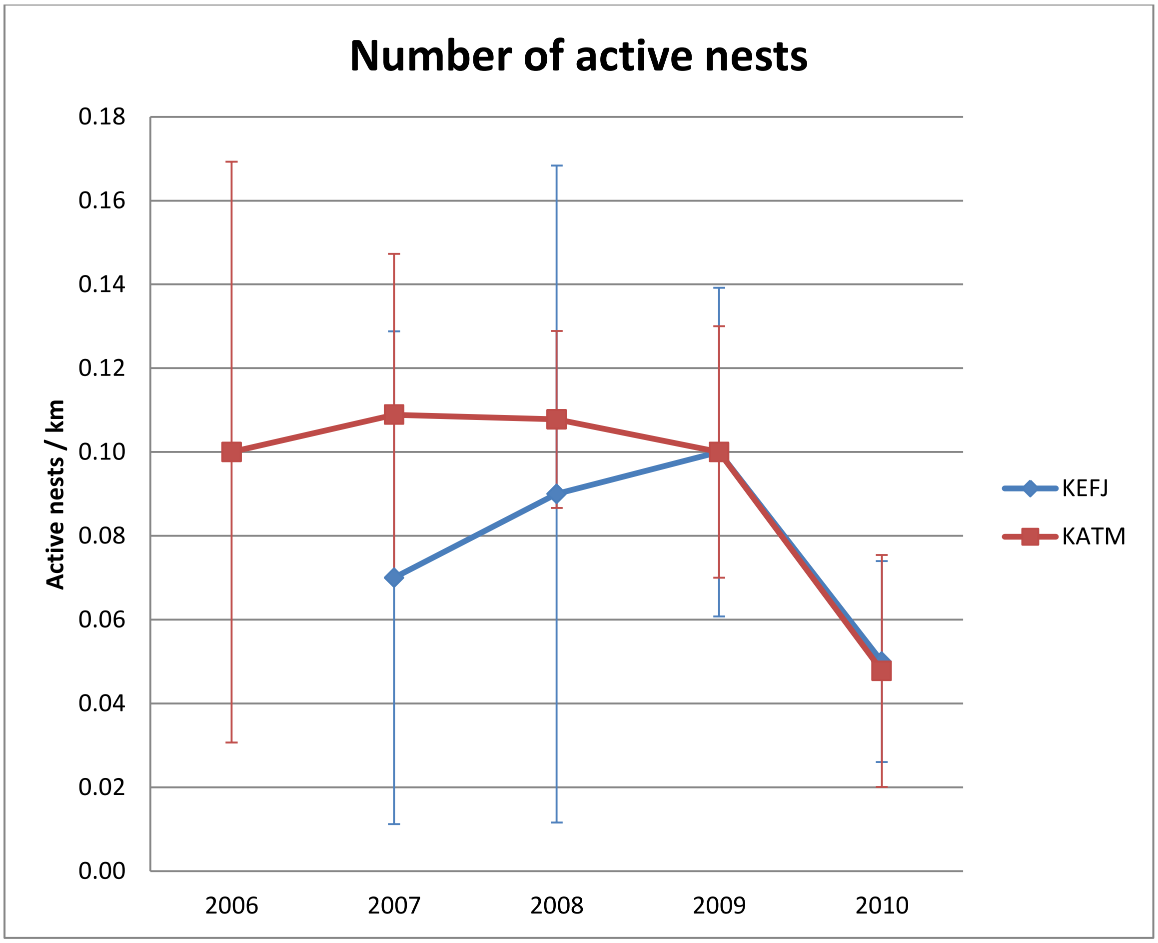


Figure 25. Number of active black oystercatcher nests / km in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

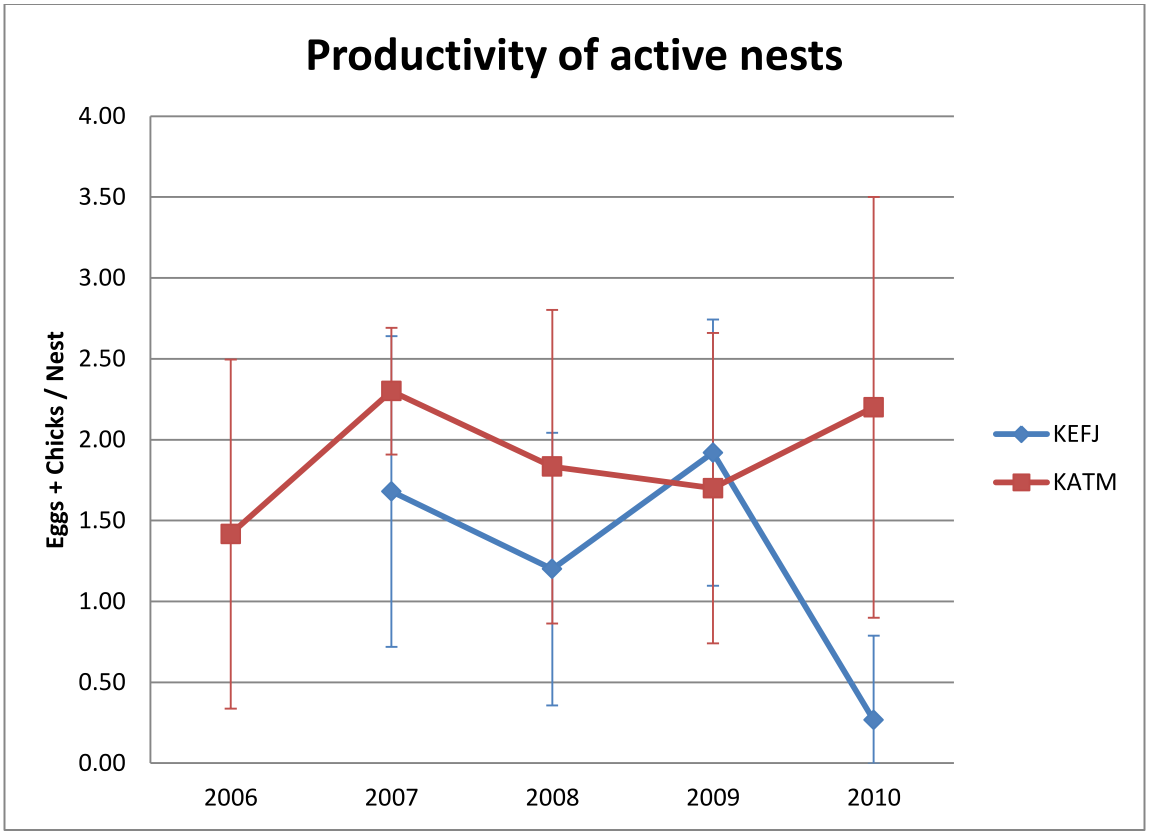


Figure 26. Productivity (egss + chicks / nest) of active black oystercatcher nests / km in KATM and KEFJ, 2006-2010. Error bars indicate 95% CI.

Diet

Three species of limpets *(Lottia pelta*, *Lottia persona*, and to a lesser extent *Lottia scutum*) and the Pacific blue mussel (*Mytilus trossulus)* were the predominant prey items found at black oystercatcher nest sites in both KATM and KEFJ (Figures 27 and 28). Together these species represented 94% of prey items found at KATM (2006-2010) nest sites and 95% in KEFJ (2007-2009) for all sampling years. No prey items were observed in KEFJ in 2010. Few nests with eggs or chicks were observed in KEFJ in 2010 and due to inclement conditions, few nests could actually be checked for the presence of prey.

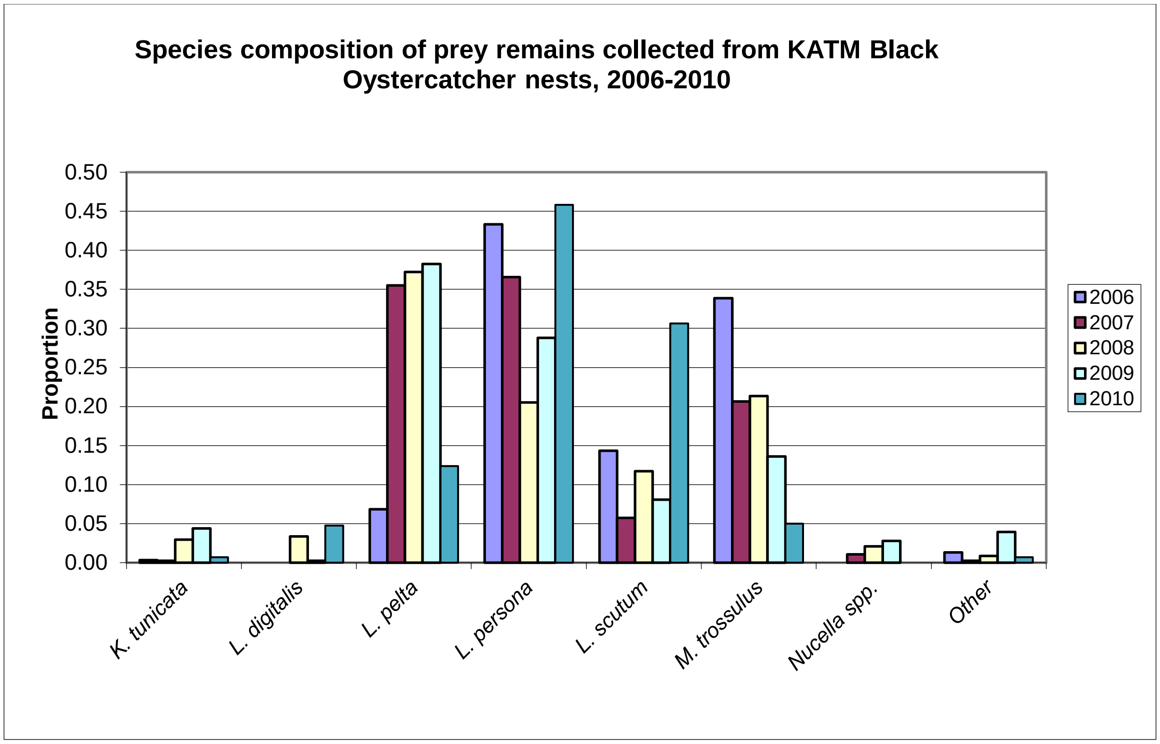


Figure 27. Species composition of prey items collected at active black oystercatcher in KATM, 2006-2010.

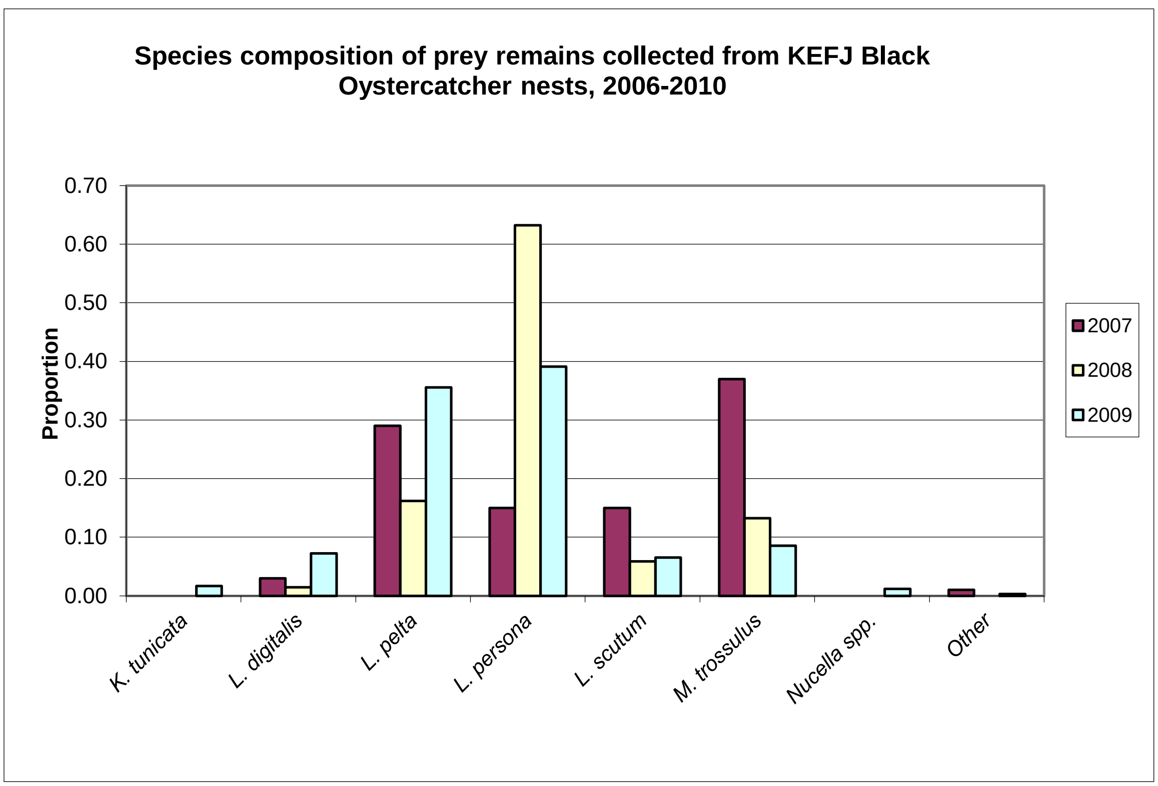


Figure 28. Species composition of prey items collected at active black oystercatcher in KEFJ, 2007-2009. No prey items were observed in 2010.

Prey size is measured for all species. However, we report only on the mean size of the most predominate limpet, *Lottia persona* and the mussel, *Mytilus trossolus*. Both of the species are also monitored for density and size within the Sampling of Intertidal Invertebrates and Algae on Sheltered Rocky Shores SOP (Dean and Bodkin 2011b). Mean *L. persona* size ranged from 18.84 to 23.02 mm in KATM from 2006-2010 and ranged from 21.20 to 22.96 mm in KEFJ from 2007-2009 (no prey items observed in 2010) (Figure 29). Mean *M. trossulus* size ranged from 27.44 to 45.05 mm in KATM from 2006-2010 and ranged from 24.45 to 29.92 mm in KEFJ from 2007-2009 (no prey items observed in 2010) (Figure 30).

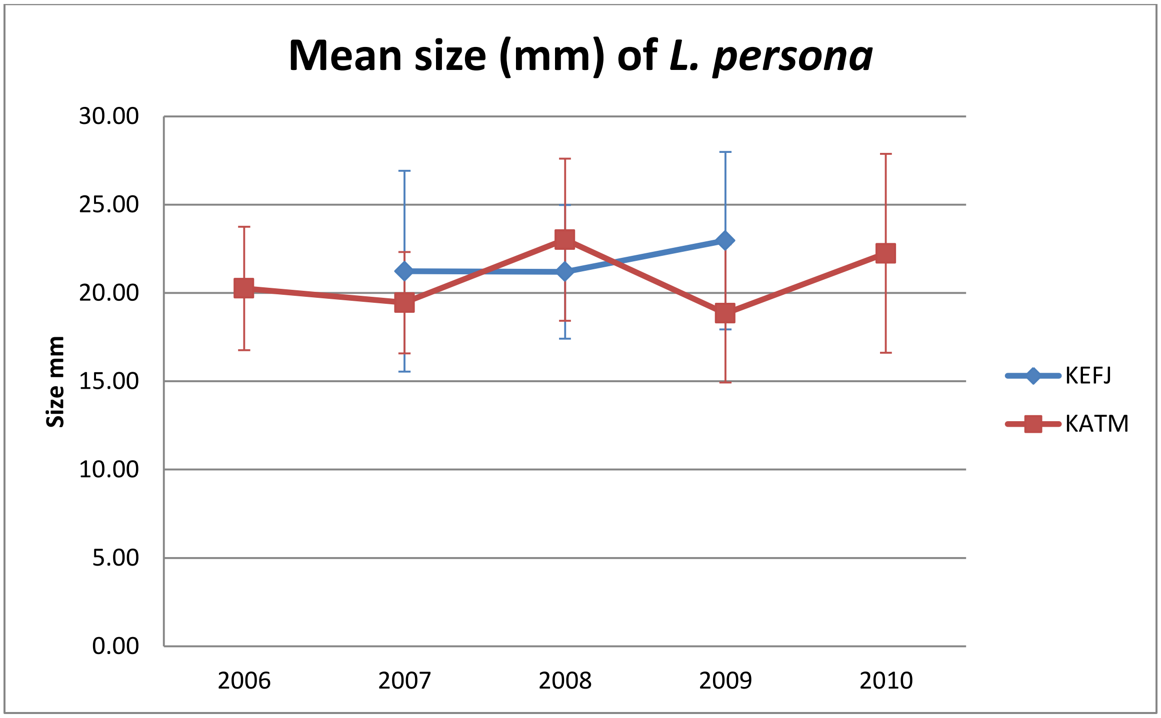


Figure 29. Mean size of *L. persona* measure at active black oystercatcher nests in KATM (2006-2010) and KEFJ (2007-2009). No prey items were observed in KEFJ in 2010.

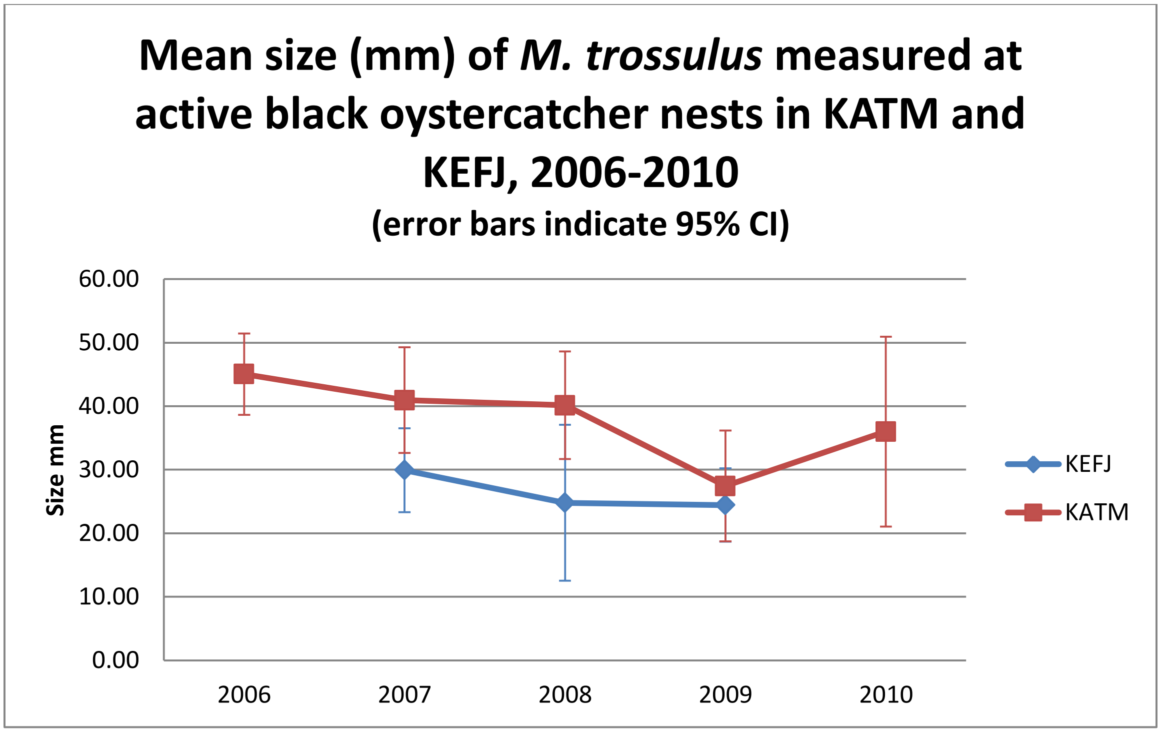


Figure 30. Mean size of *M. trossulus* measure at active black oystercatcher nests in KATM (2006-2010) and KEFJ (2007-2009). No prey items were observed in KEFJ in 2010.

Discussion

Measures of active nest density were similar again in KATM and KEFJ in 2010. Nest densities for both parks in 2010 was the lowest observed since the implementation of the monitoring protocols. Productivity of active nests was higher in KATM than KEFJ. Reasons for diminished density in both parks and low productivity in KEFJ are unclear at this point. *L. pelta* was the dominant prey item in KATM while no prey items were collected in KEFJ in 2010 due to the lack of observed chicks. In general *L. pelta* are similar in size in KEFJ and in KATM across the years, while *M. trossulus* appears larger in KATM than KEFJ, but not significantly. Our data continues to show that black oystercatchers are targeting the larger size classes of mussels and limpets, based on our random sampling in the rocky intertidal and mussel bed sites. Variation in sizes of prey was generally low. This is not surprising, but may be a key metric for monitoring purposes. Measurements of sea otter prey, pre- and post- arrival of sea otters in Glacier Bay, AK, have indicated a decline in prey sizes correlated with the increased occupation of Glacier Bay proper with sea otters (Bodkin et al. 2007a and c). A similar result may possibly occur as densities in nesting black oystercatchers changes. Lower densities of black oystercatchers may lead to increased densities of larger size classes of mussels and limpets sampled at the rocky intertidal sites and mussel beds or nest sites. The reverse may also be possible. Increased black oystercatcher densities may decrease the densities of the larger size classes of prey.

Recommendations

Surveys of black oystercatcher abundance, nest density, and diet as reflected through prey remains brought to provision chicks has been successfully implemented in KATM and KEFJ and has shown that at appropriate spatial scales of analysis, our data should continue to be collected with little revision. Sampling at the current intensity should allow us to detect trends in changes of nest density, productivity and diet (especially prey size) of the black oystercatcher. It appears as though breeding pairs may have multiple nests at a nest site and care should continue to be taken to recognize these as comprising the same nest site. It will be important to conduct future surveys as close as possible in time to these initial surveys and care must continue to be taken to minimize the disturbance to nests during sampling.

Sea Otter

Introduction

Sea otters (*Enhydra lutris*) are a common, conspicuous, and important component of the nearshore trophic food web throughout the North Pacific. They occupy all types of nearshore habitats from sheltered bays, estuaries, and fjords to exposed rocky coastlines (Kenyon 1969), but are constrained by their diving ability to habitats shallower than 100 m depth (Bodkin et al. 2004) and a near exclusive dietary reliance on benthic invertebrate prey (Riedman and Estes 1990). As a consequence of their nearshore distribution and relatively small home ranges, a rich literature exists on the biology, behavior, and ecology of the species. The sea otter provides one of the best documented examples of top-down forcing effects on the structure and function of nearshore marine ecosystems in the North Pacific Ocean (Kenyon 1969, VanBlaricom and Estes 1988, Riedman and Estes 1990, Estes and Duggins 1995) and are widely regarded as a “keystone” species in coastal marine ecosystems (Power et al. 1996). They cause well described top-down cascading effects on community structure by altering abundance of prey (e.g. sea urchins) which can in turn alter abundance of lower trophic levels (e.g. kelps). Sea otters generally have smaller home ranges than other marine mammals; eat large amounts of food; are susceptible to contaminants such as those related to oil spills; and have broad appeal to the public. From the mid-1980s through 2005 declines in sea otters have been observed in the Aleutian Islands (Doroff et al. 2003, Estes et al. 2005, Burn and Doroff 2005). As a result, the Western Alaska stock of sea otters, which occurs from Cook Inlet to the Western Aleutian Islands and includes KATM as well as Aniakchak National Monument and Preserve, was federally listed in September 2005 as threatened.

For the reasons outlined above, several metrics related to sea otters are incorporated under this vital sign. They include: observations of sea otter foraging, carcass collections to evaluate the age structure of the dying population, and aerial surveys to estimate population abundance. Because sea otters occupy areas outside the nearshore zone, aerial surveys are conducted to increase the accuracy of abundance estimates (Bodkin and Udevitz 1999).

Sea otter population abundance and trends are frequently influenced by the type and quantity of available prey (Kenyon 1969, Monson et al. 2000). Observations of foraging sea otters provide information on food habits, foraging success, (mean proportion of feeding dives that are successful) and efficiency (mean kcal/dive) based on prey numbers, types and sizes obtained by feeding animals. Because sea otter populations are often prey limited, data on foraging behavior will be useful in evaluating potential causes for differences in sea otter densities or trends among regions or years (Estes et al. 1982, 2003b, Gelatt et al. 2002, Dean et al. 2002, Bodkin et al. 2002, Tinker et al. 2008).

Due to high spatial variability in marine invertebrate populations (e.g. extreme patchiness) and difficulty in sampling underwater prey populations, foraging sea otters provide an alternative method to direct sampling of subtidal invertebrates. Following a successful foraging dive, sea otters return to the surface to consume their prey. This provides the opportunity to identify, enumerate, and estimate the size of the benthic organisms they consume. Therefore sea otter foraging observations will provide data on species composition and sizes of subtidal invertebrate prey populations that are difficult to obtain directly. Observations collected over time may allow inference to changes in the species composition and sizes of the nearshore benthic invertebrate communities.

As a result of their nearshore distribution and relatively high density, moribund sea otters often haul out ashore, or their carcasses drift onto beaches. Annual collections of sea otter carcasses provide a record of the ages of dying individuals through analysis of dentin deposition in teeth (Bodkin et al. 1997). The age distributions of dying sea otters generated from annual carcass collections can provide a baseline against which future distributions can be compared and potentially provide inference regarding causes for change in population abundance , behavior, or diet (Monson et al. 2000, Estes et al. 2003a). Combined with data from a fresh carcass stranding program or annual population surveys, age-specific mortality data modeling can be used to inform managers regarding conservation decisions related to causes of mortality (Gerber et al. 2004, Tinker et al. 2006).

Methods

Prey composition, foraging success rate, and prey size were obtained from shore based observations of randomly selected foraging otters. Shore-based observations limited data collection to sea otters feeding within approximately 1 km of shore. High powered telescopes (Questar Corp., Hew Hope, PA.) and 10X binoculars were used to record prey type, number, and size class during foraging bouts of focal animals. A bout consisted of observations of repeated dives for a focal animal while it remains in view and continues to forage (Calkins 1978). Assuming each foraging bout records the feeding activity of a unique individual, bouts were considered independent while dives within bouts were not. Thus the length of any one foraging bout was limited to 20 dives, or one hour, after which a new focal animal was chosen. Within each bout sampled the following metadata were recorded: date, start and end time, ageclass, sex, reproductive status of the individual and location coordinates. Foraging data collected include dive and inter-dive times, success, prey species, number and size, and if prey were given or taken (typically given to a pup, or taken by a con-specific). The sampling design included the acquisition of foraging data within a 10 km radius of each of the five established rocky intertidal invertebrate and algal sites. The objective was to annually obtain data from 10 individuals within each of these 10 km buffers, a total of 50 bouts per year.

Sea otters in the study areas were generally not individually identifiable. In addition, some foraging areas may have been used more than others by individuals and by otters living in the area in general. Therefore individual sea otters may have been observed more than once leading to potential bias toward individuals sampled more than once. To minimize this potential, observers use characteristics such as sex, sizes, coloration, and reproductive status to identify individuals. If more than one animal was observed foraging, selection was based on proximity, alternating between closest and furthest.

Throughout the study areas in KATM and KEFJ, we have identified segments of shoreline or offshore islands to search for sea otter carcasses. These areas have been consistently searched by two or more observers. Search patterns cover from the storm strand line to the water's edge and focus on areas where larger amounts of debris collect. When a carcass is encountered the skull and baculum, if present, are collected. The following data are recorded: date, observers, condition of carcass, parts collected, latitude/longitude, location on beach (e.g. strand line, above high tide, etc), and cause of mortality (usually not known). A premolar tooth (or substitute if the premolar is not available) is sent to Matson's Laboratory in Montana for cementum layer age analysis.

Of the various metrics measured in regards to the sea otter vital sign, only foraging observations and carcass collections have been collected in KATM since 2006 and KEFJ since 2007. Here we will be reporting only on the descriptive analyses associated with data acquired directly from observations of foraging sea otters and age-specific mortality. Results from aerial surveys conducted in KATM in 2008 and in KEFJ in 2007 and 2010are reported elsewhere (Bodkin et al 2008, Coletti et al. 2009 and 2011).

One of the objectives for this monitoring program is to detect levels of change deemed ecologically important (Dean and Bodkin 2011a). For the sea otter foraging data we have established a 0.35 change in the proportion of dominant prey categories, a 0.50 change in prey size and a 0.20 increase or 0.33 decrease in the number of hours needed to meet energetic requirements as ecologically relevant changes to detect. Programming capable of providing variance estimates of energy recovery rates is presently in revision, precluding power analysis for this metric. Power analysis for liner regression (Gerrodette 1993) was used to evaluate levels of change in focal species densities that could be detected over time. Forage data are analyzed at the spatial scale of a park. Future analyses may include finer spatial resolution analyses as sample sizes increase within each of the five buffers associated with the intertidal sites and should include caloric recovery rate power analyses.

Results

During five field seasons (2006-2010) at KATM we obtained data from 234 independent sea otter foraging bouts, consisting of 2,222 dives (Table 2). The prey recovery success rate was 90% for dives with known outcomes (range 87% - 93%) (Figure 31). During four field seasons (2007-2010) at KEFJ we obtained data from 195 independent sea otter foraging bouts, consisting of 1,627 dives (Table 2). The prey recovery success rate was 86% for dives with known outcomes (range 68% - 96%) (Figure 31).

Table 2. Summary of sea otter foraging observations in KATM and KEFJ from nearshore monitoring data collection, 2006 - 2010. A bout is the sampling unit for data analysis**.**

Year	Number of bouts observed		Number of dives observed		Mean number of observed dives per bout		St. error number of dives per bout	
	KATM	KEFJ	KATM	KEFJ	KATM	KEFJ	KATM	KEFJ
2006	65	.	451	.	6.74	.	0.24	.
2007	54	44	498	470	7.66	8.91	0.24	0.31
2008	38	57	427	392	8.57	5.73	0.28	0.23
2009	36	37	392	269	8.43	7.16	0.29	0.34
2010	41	57	454	496	7.83	6.96	0.25	0.25
All Years	234	195	2,222	1,627	7.82	7.26	0.12	0.14

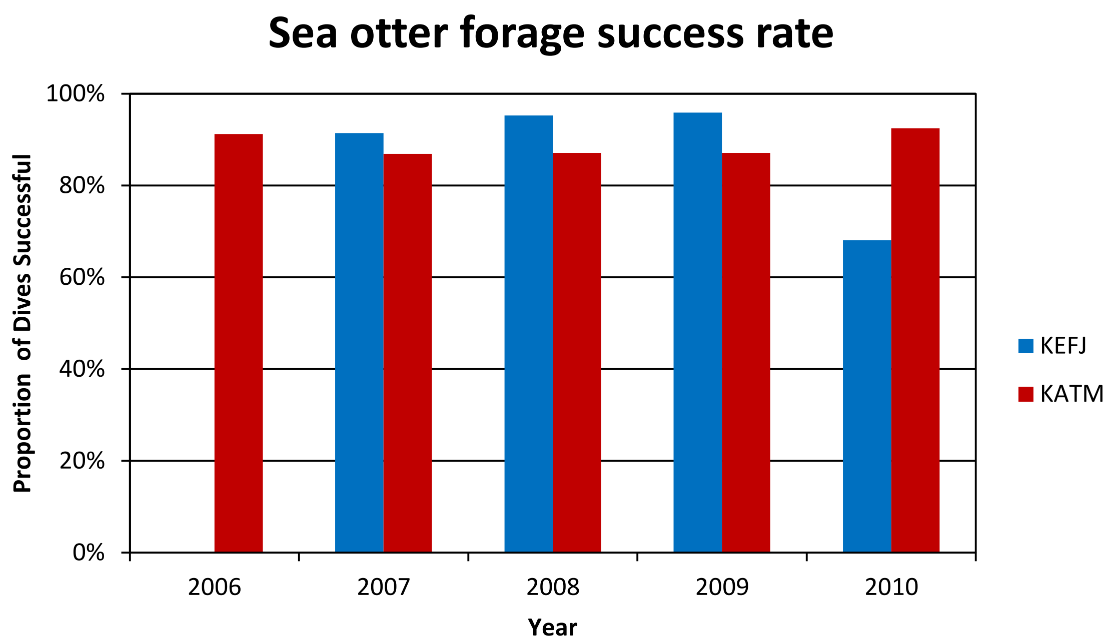
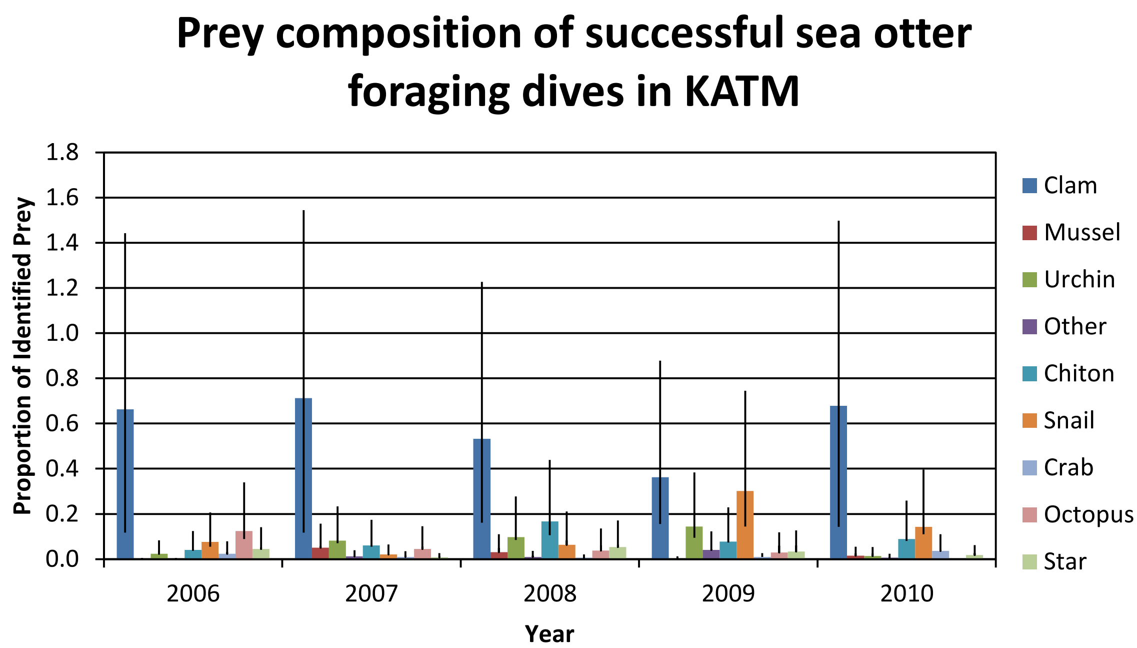


Figure 31. Proportion of known outcome dives where prey was successfully retrieved by foraging sea otters in KEFJ and KATM, 2006-2010. Data collection began in KEFJ in 2007. Dives in which otters were retrieving a previously collected prey item that had been dropped were not included. Additionally, a dive is only counted towards the success rate once, even if more than 1 item was retrieved.

Since 2006, we have observed sea otters feeding on at least 30 different prey items including bivalves, decapod crustaceans, gastropods, and echinoderms (Table 3). At KATM, clams dominated sea otter diets across all years of data collection, comprising greater than 60% of the diet (Figure 32). In 2006 octopus accounted for 12% of identified prey, in 2008 chitons were 16%, and in 2009 snails and urchins accounted for 30% and 14%, respectively. Otherwise, chitons, crabs, mussels, octopus, snails, sea stars, sea urchins, and other prey each comprised less than 10% of the of prey recovered. At KEFJ, mussels (*Mytilus trossulus*) dominated sea otter diets across all years of data collection, comprising 65% of the diet (Figure 32). In all years, clams were the second most prominent prey item comprising 23% of the diet. Otherwise, chitons, crabs, octopus, snails, sea stars, sea urchins, and other prey each comprised less than 10% of the of prey recovered. Annually there has been little observed change in the predominant prey category at either Park.

Table 3. List of prey items that sea otters were observed consuming in KATM and KEFJ, 2006 - 2010.

Phylum (Subphylum)	Class (Order)	Prey Item (Genus, species)
Mollusca	Polyplacaphora	*Cryptochiton stelleri*, *Katharina tunicata*
	Gastropod	*Neptunea* spp., *Nucella* spp.
	Bivalvia	*Macoma nasuta*, *Macoma* spp., *Mya truncata*, *Mya* spp., *Leukoma staminea* (formerly *Protothaca staminea*), *Saxidomus gigantea*, *Clinocardium nutallii*, *Modiolus modiolus*, *Mytilus Trossulus* (also gravid *M. trossulus*), *Pododesmus macroschisma*, *Chlamys* spp., *Tresus capax*
	Cephalopoda	*Octopus dofleini*
Echiura		*Echiurus* spp.
Arthropoda (Crustacea)		
	Cirripedia (Decapoda)	*Cancer* spp., *Telmessus cheiragonus*, *Pagurus* spp.
Echinodermata		
	Asteroidea	*Solaster* spp., *Pisaster* spp.
	Ophiuroidea	*Ophiurid* spp.
	Echinoidea	*Strongylocentrotus droebachiensis*, *S. purpuratus*, *Dendraster excentricus*
	Holothuroidea	*Cucumaria fallax*
Chordata	Chondrichthyes	skate egg case



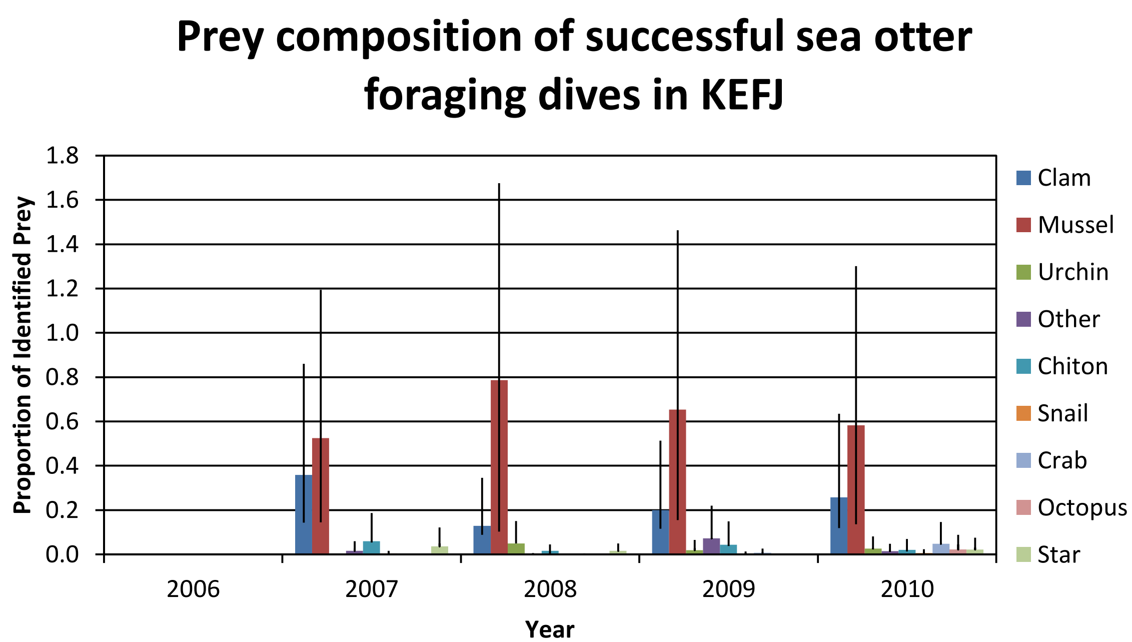


Figure 32. Proportion of identified prey retrieved by foraging sea otters in KATM and KEFJ from 2006 through 2010. The top graph is data from KATM, bottom is from KEFJ. Data collection began in KEFJ in 2007. Unidentified prey items are not included in these calculations. Additionally, a prey item is only counted towards the proportion once, even if more than 1 of the same item was retrieved on the same dive. Error bars represent 95% confidence intervals.

Sizes of prey captured by foraging sea otters vary by species (Figure 33). In KATM, the predominant prey, clams, averaged 56 mm over all sites and all years combined. Crabs, snails, mussels, urchins, and unidentified prey items were smaller than the clams being retrieved while chitons, stars, and other prey items (e.g. sea cucumbers, marine worms) were larger than the clams. It is difficult to estimate a mean size for the octopus we observed the otters eating. All were larger than our largest estimated size class (based on paw width of an 'average' sea otter) of 156 mm. We field estimated mass for 60% of the observed octopus at 5.8 kg.

In KEFJ, the predominant prey, mussels, averaged 25 mm over all sites and all years combined and this size was consistent across years. Clams averaged 53 mm and unidentified prey items were 29 mm. Sample sizes were low for the other prey categories, but their averages are reported in Figure 33.

For prey where we have more than ten size observations per year, mean size per year has been reported in Figure 34. There is no observed difference in the size of clams between Parks nor across years. Mussels, the primary prey item in KEFJ, have a similar mean size across years in KEFJ. Data are too scant to determine if the larger average size at KATM is meaningful. Unidentified prey size at both Parks does not vary much across years. In KATM unidentified prey are consistently smaller than the mean size of the predominant prey while in KEFJ unidentified prey are similar in size to the predominant prey.

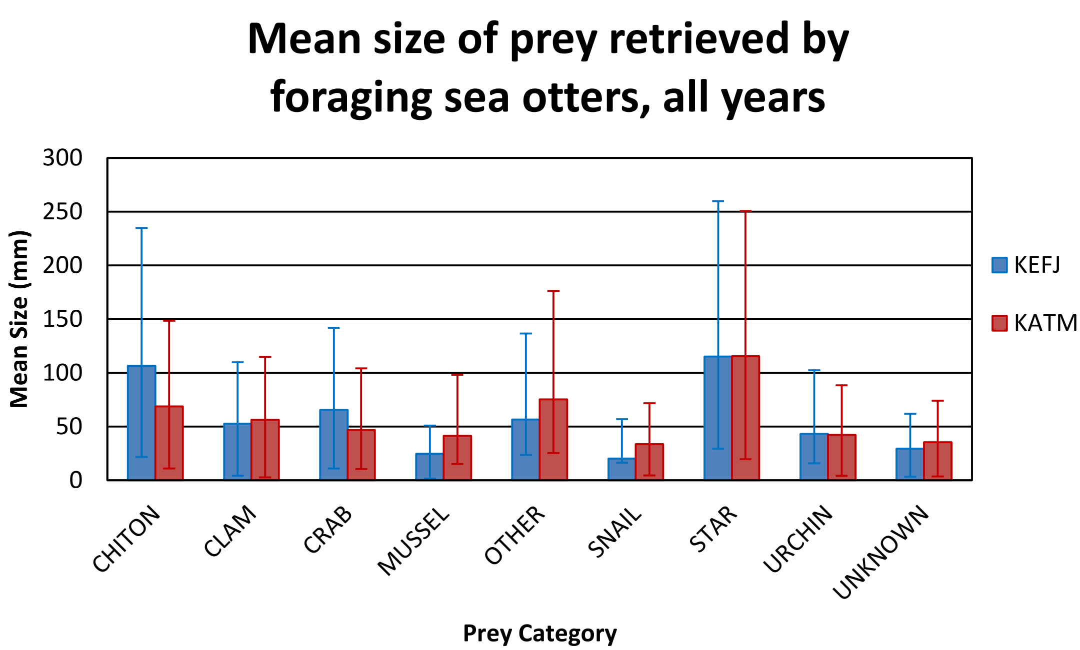
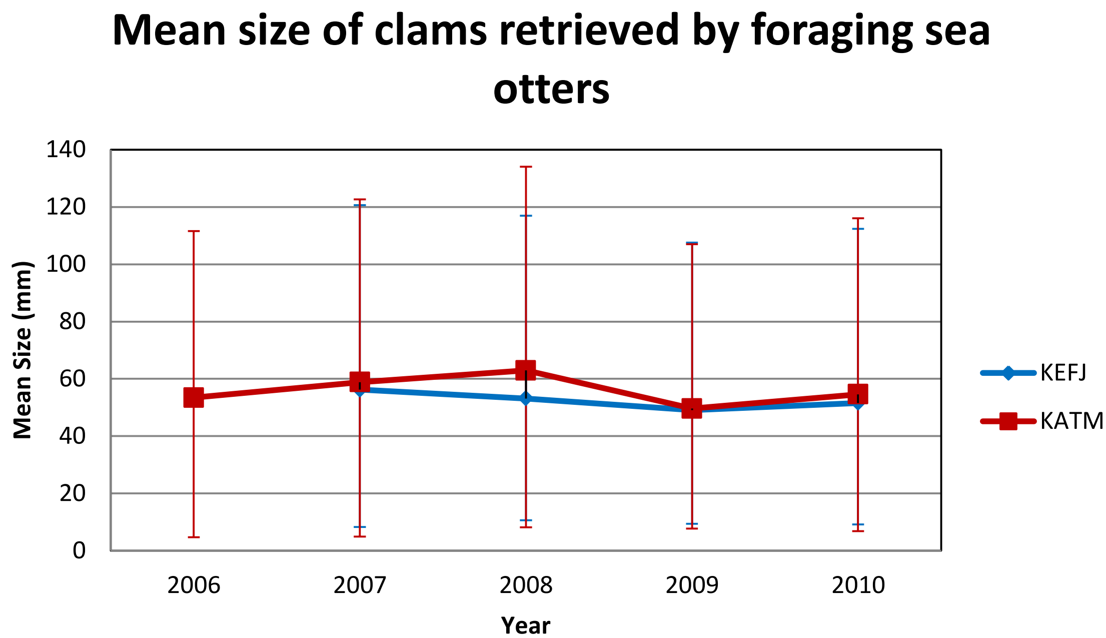
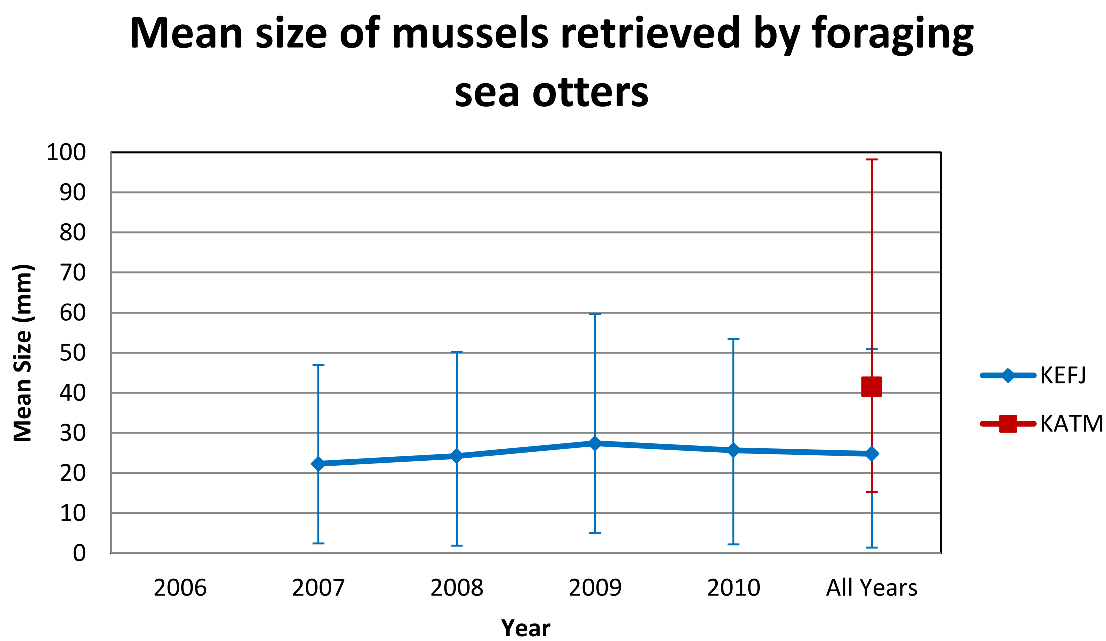


Figure 33. Mean size of prey items recovered by prey category by foraging sea otters in KATM (2006-2010) and KEFJ (2007-2010). Sizes from all prey items retrieved were used in the calculations. Error

bars represent 95% confidence intervals. The octopus category was excluded from this figure because methods of assessing size have not been standardized for larger prey items.





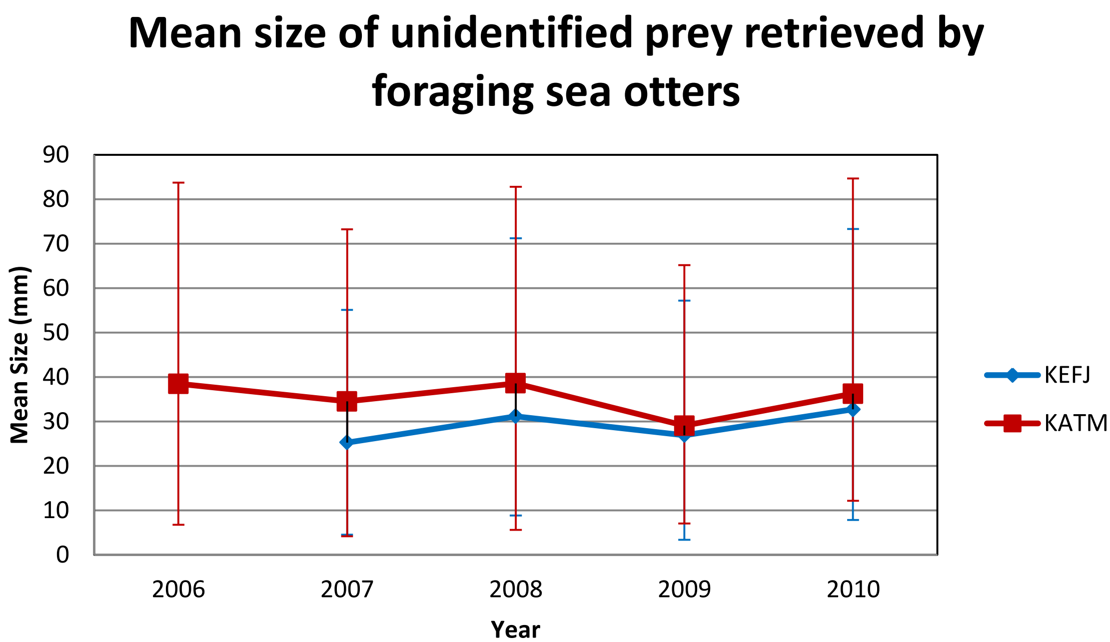


Figure 34. Mean size of specific prey items recovered by prey category by foraging sea otters in KATM and KEFJ, by year. The top graph is clam size data from KATM, middle is mussel size data, and bottom is size data for unidentified prey items. For most prey types there were too few observations per year to graph separately. Here, the predominant prey category per region is shown. KATM had too few mussel size observations to graph by year, therefore all years were lumped for comparison to KEFJ. Error bars represent 95% confidence intervals.

Since 2006, 153 sea otter carcasses have been successfully aged from KATM. The proportion of carcasses in each age bin from the five collection years combined are shown in Figure 35. The proportion of carcasses in each age bin from each of the five collection years are shown in Figure 36.

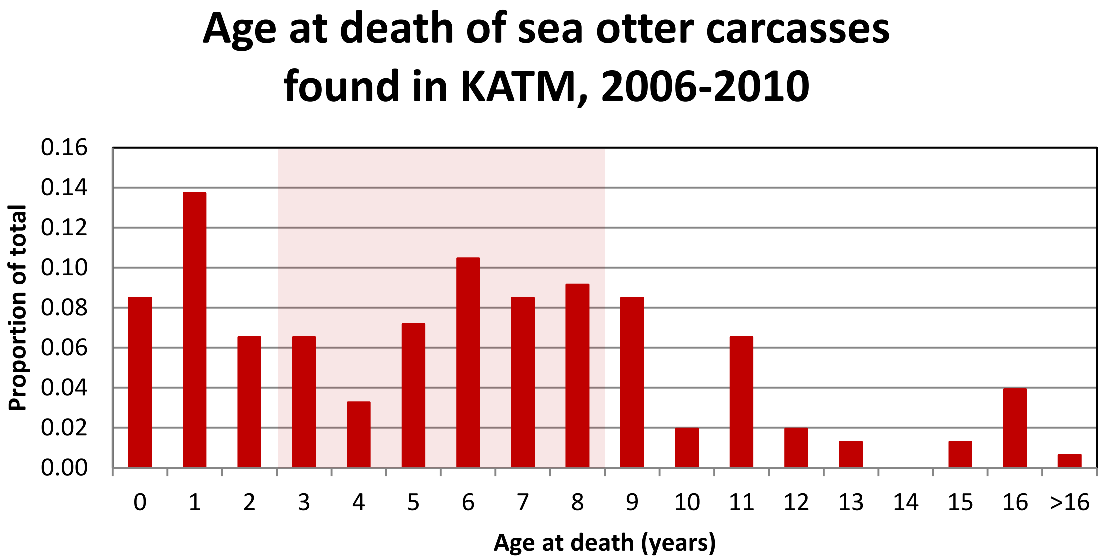


Figure 35. Age at death of sea otter carcasses found beachcast in KATM from 2006 through 2010. Age was determined using tooth cementum analysis. The shaded area represents ages considered prime reproductive ages. To date, there are not enough recovered carcasses from KEFJ to plot.

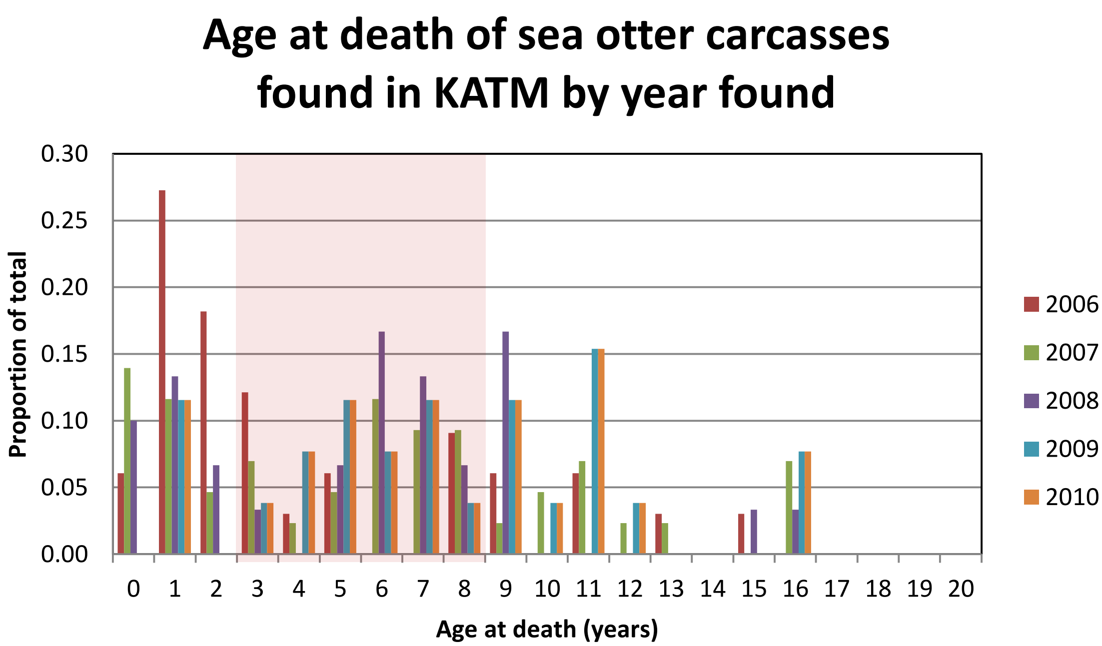


Figure 36. Age at death of sea otter carcasses found beachcast in KATM per year from 2006 through 2010. Age was determined using tooth cementum analysis. The shaded area represents ages considered prime reproductive ages.

Discussion

Using the methods briefly described above, we were able to estimate sea otter foraging success, prey composition, mean prey size, and age-specific mortality. Predominant prey varied between parks, but within each park was consistent over time. We anticipate that further development of the model to analyze rates of energy recovery will allow us to detect ecologically meaningful levels of change in the future. Foraging success rates were similar across years and between parks, except for the low rate in KEFJ in 2010. This appears to be due to the inclusion of a higher than typical number of bouts from juvenile otters. Juveniles are hypothesized to be still acquiring the necessary skills for successful independent foraging and have been observed in other areas to have lower success rates than adults. Subsequent analyses will account for ageclass. Overall a wide range of prey items was observed in both parks. Sea otters display individual preferences in prey selection that can be attributed to prey availability, maternally derived learning and likely several other factors. Since this monitoring protocol has no plans for marking and following individual sea otters' dietary preferences, our analyses will focus on population-level metrics that can be compared over time and to other populations. In KATM, the primary prey category across years is clams, while in KEFJ, mussels predominate. Unidentified prey is a large component of the diet in both parks. Our developing forage model addresses the unidentified prey component by resampling the known items weighting for other known metrics such as retrieval time, consumption time, and size. Analyses are underway to determine if our methods will allow detection of the levels of change deemed ecologically important. Power analyses from past sea otter studies indicate that we will be able to do so. In the course of collecting the observations we have determined the need to address larger prey items such as octopus and fish. Our methods allow us to estimate the size of a prey item by comparing the item to the otter's paw width. In other research otter paws have been measured yielding a mean otter paw width of 52 mm. This method is successfully employed by sea otter researchers throughout the sea otter range; however it is proving difficult to adapt to extremely large items that exceed 4 paw widths. We are working to develop alternate methods of sizing these items and there is already a mechanism to include them in the forage data model.

Searches for sea otter carcasses continue in KEFJ and KATM. To date, we have not recovered sufficient carcasses from KEFJ to employ age-specific mortality analyses. Discussions are underway to determine ways to improve our carcass recoveries in KEFJ such as adding areas of shoreline to search or searching more frequently to recover carcasses prior to removal by scavengers. Descriptive analysis of the KATM data has yielded several questions that need to be addressed. There appears to be a higher than expected proportion of prime age animals but it is not yet certain if this is true across all years. While this may be due to oversampling of male haul outs, this is similar to observations from areas that experienced known sea otter food-limited population crashes. We have age at death datasets from several areas with different known population statuses (e.g. at equilibrium, before and during a population crash) that can be used for comparisons with this data. We are currently working on a resampling exercise using likelihood based tests to examine whether age-specific survival is stable over time.

Recommendations

Based on these results, we recommend continued collection of sea otter foraging data with an emphasis on completing the analysis model. Additionally, 50 bouts should be set as the minimum target. Results should be viewed both longitudinally and within the larger framework of known otter foraging studies for context. Sea otter carcass collections should also be continued and the expansion of collection efforts should be seriously considered for both parks. It will be important to build an analysis model that facilitates the inclusion of additional data over time to recognize emerging trends.

Literature Cited

Alaska Shorebird Working Group. 2000. Alaska shorebird conservation plan. Unpublished Report, U. S. Fish and Wildlife Service, Anchorage, Alaska. 47 pp.

Andres, B. and G. A. Falxa. 1995. Black Oystercatcher (*Haematopus bachmani*) In The Birds of North America, No. 155 The Academy of Natural Sciences, Philadelphia, and The American Ornithologists' Union, Washington, D.C., Philadelphia, PA.

Andres, B. A. 1998. Shoreline habitat use of Black Oystercatchers breeding in Prince William Sound, Alaska. Journal of Field Ornithology 69(4): 626-634.

Andres, B. A. 1999. Effects of persistent shoreline oil on breeding success and chick growth in black oystercatchers. The Auk 116:640-650.

Barry, J. P., C. H. Baxter, R. D. Sagarin, and S. E. Gilman. 1995. Climate-related, long-term faunal changes in a California rocky intertidal community. Science 267(5198): 672-675.

Bodkin JL, Ames JA, Jameson RJ, Johnson AM, Matson GE. 1997. Estimating age of sea otters with cementum layers in the first premolar. J Wildl Manage 61(3):967–973.

Bodkin, J. L., and M. S. Udevitz. 1999. An aerial survey method to estimate sea otter abundanceG. W. Garner, S. C. Amstrup, J. L. Laake, B. F. J. Manly, L. L. McDonald, and D. G. Robertson. Marine Mammal Survey and Assessment Methods. A.A. Balkema, Rotterdam.

Bodkin, J.L., B. Ballachey, T.A. Dean, F.K. Fukuyama, S.C. Jewett, L.L. McDonald, D.H. Monson, C.E. O'Clair, and G.R. VanBlaricom. 2002. Sea otter population status and the process of recovery following the 1989 Exxon Valdez oil spill. Marine Ecology Progress Series 241:237-253.

Bodkin, J. L., G. G. Esslinger, and D. H. Monson. 2004. Foraging depths of sea otters and implications to coastal marine communities Marine Mammal Science 20(2):305-321.

Bodkin, J.L., D.H. Monson, and G.G. Esslinger. 2007a. Population status and activity budgets derived from time-depth recorders in a diving mammal. J. Wildlife Management 71(6):2034-2044.

Bodkin, J. L., T. A. Dean, and H. A. Coletti. 2007b. Nearshore Marine Monitoring in the Southwest Alaska Network of National Parks. National Park Service. Anchorage, AK. 102 pg.

Bodkin, J.L., B.E. Ballachey, G.G. Esslinger, K.A. Kloecker, D.H, Monson, and H.A. Coletti. 2007c. Perspectives on an invading predator—Sea otters in Glacier Bay, *in* Piatt, J.F., and Gende, S.M., eds., Proceedings of the Fourth Glacier Bay Science Symposium, October 26–28, 2004: U.S. Geological Survey Scientific Investigations Report 2007-5047, p. 133-136.

Bodkin, J.L., T.A. Dean, H.A. Coletti, and K.A. Kloecker. 2008. Nearshore marine vital signs monitoring in the Southwest Alaska network of National Parks. Annual report to the US National Park Service, Anchorage, AK. 176 pp.

Bodkin, J. L. 2001a. SOP for conducting marine bird and mammal surveys - Version 4.1: Southwest Alaska Inventory and Monitoring Network. Natural Resource Report NPS/SWAN/NRR—2011/392. National Park Service, Fort Collins, Colorado.

Bodkin, J. L. 2011b. SOP for monitoring black oystercatchers - Version 1.1: Southwest Alaska Inventory and Monitoring Network. Natural Resource Report NPS/SWAN/NRR—2011/391. National Park Service, Fort Collins, Colorado.

Bostrom, C., Jackson, E.L., Simenstad, C.A., 2006. Seagrass landscapes and their effects on associated fauna: a review. Estuar. Coast. Shelf S. 68, 383–403.

Brown, S., C. Hickey, B. Harrington, and R. Gill. 2001. The U.S. Shorebird Conservation Plan, 2nd ed. Brown, S., Hickey, C., Harrington, B., Gill, R. eds. Brown Manomet Center for Conservation Sciences, Manomet, MA.

Burn, D.M. and A.M. Doroff. 2005. Decline in sea otter (*Enhydra lutris*) populations along the Alaska Peninsula, 1986-2001. Fishery Bulletin 103:270-279.

Calkins DG (1978) Feeding behavior and major prey species of the sea otter *Enhydra lutris* in Montague Straits, Prince William Sound, Alaska. Fish Bull 76:125–131

Coletti, H., J. Bodkin, T. Dean, and K. Kloecker. 2009. Nearshore marine vital signs monitoring in the Southwest Alaska Network of National Parks. Natural Resource Technical Report NPS/SWAN/NRTR—2009/252. National Park Service, Fort Collins, Colorado.

Coletti, H. A., J. L. Bodkin, and G. G. Esslinger. 2011. Sea otter abundance in Kenai Fjords national Park: results from the 2010 aerial survey: Southwest Alaska Inventory and Monitoring. Natural Resource Technical Report NPS/SWAN/NRTR—2011/417. National Park Service, Fort Collins, Colorado.

Dayton, P. K. 1971. Competition, disturbance and community organization: the provision and subsequent utilization of space in a rocky intertidal community. Ecological Monographs 41(4): 351-89.

Dean, T.A. M.S. Stekoll, S.C. Jewett, R.O. Smith, J.E. Hose. 1998. Eelgrass (Zostera marina L.) in Prince William Sound, Alaska: Effects of the Exxon Valdez oil spill. Mar Pollut Bull 36:201-210

Dean T.A., L. Haldorson, D.R. Laur, S.C. Jewett, A. Blanchard. 2000. The distribution of nearshore fishes in kelp and eelgrass communities in Prince William Sound, Alaska: associations with vegetation and physical habitat characteristics. Environmental Biology of Fishes 57: 271-287.

Dean, T. A., J. L. Bodkin, A. K. Fukuyama, S. C. Jewett, D. H. Monson, C. E. O'Clair, and G. R. VanBlaricom. 2002. Sea Otter (*Enhydra lutris)* Perspective: Mechanisms of Impact and Potential Recovery of Nearshore Vertebrate Predators following the 1989 Exxon Valdez Oil Spill: Part B. Food Limitation and the Recovery of Sea Otters in Prince William Sound. Mar. Ecol. Prog. Ser. 241: 255-270.

Dean, T. A. and J. L. Bodkin. 2011a. Protocol narrative for marine nearshore ecosystem monitoring in the Southwest Alaska Network of National Parks. Natural Resource Report NPS/SWAN/NRR—2011/449. National Park Service, Fort Collins, Colorado.

Dean, T. A. and J. L. Bodkin. 2011b. SOP for sampling of intertidal invertebrates and algae on sheltered rocky shores - Version 4.6: Southwest Alaska Inventory and Monitoring Network. Natural Resource Report NPS/SWAN/NRR—2011/397. National Park Service, Fort Collins, Colorado.

Doroff, A.M., J.A. Estes, M.T. Tinker, D.M. Burn, and T.J. Evans. 2003. Sea otter population declines in the Aleutian archipelago. Journal of Mammalogy 84(1):55-64.

Draulans, D. 1982. Foraging and Size Selection of Mussels by the Tufted Duck, *Aythya fuligula*. Journal of Animal Ecology 51(3): 943-956..

Duarte, C.M., 2002. The future of seagrass meadows. Environ. Conserv. 29, 192–206.

Esler, D., J. A. Schmutz, R. L. Jarvis and D. M. Mulcahy. 2000a. Winter survival of adult female harlequin ducks in relation to history of contamination by the Exxon Valdez oil spill. Journal of Wildlife Management 64(3): 839-847.

Esler D., T.D. Bowman, C.E. O'Clair, T.A. Dean, L.L. McDonald. 2000b. Densities of Barrow's Goldeneyes during winter in Prince William Sound, Alaska, in relation to habitat, food, and history of oil contamination. Water Birds 23:423-429

Esler, D., T. D. Bowman, K. A. Trust, B. E. Ballachey, T. A. Dean, S. C. Jewett, and C. O. O'Clair. 2002. Harlequin Duck Population Recovery Following the Exxon Valdez Oil Spill: Progress, Process, and Constraints. Marine Ecology Progress Series 241:271-286.

Estes, J.A, R.J. Jameson and E.B. Rhode. 1982. Activity and Prey Election in the Sea Otter: Influence of Population Status on Community Structure. The American Naturalist, Vol. 120, No. 2, pp. 242-258.

Estes, J. A. and D. O. Duggins. 1995. Sea otters and kelp forests in Alaska: generality and variation in a community ecology paradigm. Ecological Monographs 65:75-100.

Estes, J.A., B.B. Hatfield, K. Ralls, and J. Ames. 2003a. Causes of mortality in California sea otters during periods of population growth and decline. Marine Mammal Science 19(1):198-216.

Estes, J.A., M.L. Riedman, M.M. Staedler, M.T. Tinker, and B.E. Lyon. 2003b. Individual variation in prey selection by sea otters: patterns, causes, and implications. Journal of Animal Ecology 72:144-155.

Estes, J.A., M.T. Tinker, A.M. Doroff, and D.M. Burn. 2005. Continuing sea otter population declines in the Aleutian archipelago. Marine Mammal Science. 21:169-172.

Faxla, G.A., 1992. Prey choice and habitat use by foraging black oystercatchers. Unpublished

Ph.D. dissertation, University of California, Davis.

Frank, P. W. 1982. Effects of Winter Feeding on Limpets by Black Oystercatchers, *Haematopus bachmani*. Ecology 63(5): 1352-1362.

Gelatt, T. S., D. B. Siniff and J. A. Estes. 2002. Activity patterns and time budgets of the declining sea otter population at Amchitka Island, Alaska. Journal of Wildlife Management 66(1): 29-39.

Gerber, L.H., M.T.Tinker, D.F. Doak, J.A. Estes, and D.A. Jessup. 2004. Mortality sensitivity in life-stage simulation analysis: a case study of Southern sea otters. Ecological Applications 14(5):1554-1565.

Gerrodette, T. 1993. Trends: software for a power analysis of linear regression. Wildlife Society Bulletin 21: 515-516.

Groves, S. 1984. Chick growth, sibling rivalry, and chick production in American black oystercatchers. The Auk 101: 525-531.

Hahn, T. and M. Denny. 1989. Tenacity-mediated selective predation by oystercatchers on intertidal limpets and its role in maintaining habitat partitioning by *Collisella scabra* and *Lottia digitalis*. Marine ecology progress series. Oldendorf 53(1): 1-10.

Hamilton, D., and T. Nudds. 2003. Effects of predation by common eiders (*Somateria mollissima*) in an intertidal rockweed bed relative to an adjacent mussel bed. Marine Biology 142(1): 1-12.

Hartwick, E. B. 1976. Foraging of the black oyster catcher (Haematopus bachmani Audubon). Canadian Journal of Zoology 54(2): 142-155.

Hauxwell, J., J. Cerbian, and I Valiela. 2003. Eelgrass Zostera marina loss in temperate estuaries: Relationship to land derived nitrogen loads and effect of light limitation by algae. Mar. Ecol. Prog. Ser. 247:59-73.

Hazlitt, S. and R. W. Butler. 2001. Site fidelity and reproductive success of black oystercatchers in British Columbia. Waterbirds 24(2): 203-207.

Irons, D. B., D. R. Nysewander and J. L. Trapp. 1988. Prince William Sound Sea Otter Distribution in Respect to Population Growth and Habitat Type. Prepared by Alaska Investigations Field Office (Branch of Wetlands and Marine Ecology) and Wildlife Assistance (Marine Bird Project).

Irons, D.B., S.J. Kendall, W.P. Erickson, L.L. McDonald, and B.K. Lance. 2000. Nine years of Exxon Valdez oil spill. Effects of marine birds in Prince William Sound. Condor:120:723-737.

Jamieson, G. S., E. D. Grosholz, D. A. Armstrong, and R. W. Elner. 1998. Potential Ecological Implications from the Introduction of the European Green Crab, *Carcinus maenas* (Linneaus), to British Columbia, Canada, and Washington, USA. Journal of Natural History 32(10-11):

1587-1598.

Jewett, S.C., T.A. Dean, R.O. Smith, A. Blanchard. 1999. The *Exxon Valdez* oil spill: Impacts and recovery in the soft-bottom benthic community in and adjacent to eelgrass beds. Mar Ecol Prog Ser 185:59-83.

Keough, M. J., G. P. Quinn and A. King. 1993. Correlations between human collection and intertidal mollusk populations on rocky shores. Conservation Biology 7(2): 378-390.

Keough, M. J. and G. P. Quinn. 1998. Effects of periodic disturbances from trampling on rocky intertidal algal beds. Ecological Applications 8(1): 141-161.

Kenyon, K. W. 1969. N. American Fauna 68: The Sea Otter in the Eastern Pacific Ocean. US Fish and Wildlife Service. pp. 352.

Kingsford, M. J., A. J. Underwood and S. J. Kennelly. 1991. Humans as predators on rocky reefs in New South Wales, Australia. Marine Ecology Progress Series. Oldenforf 72(102): 1-14.

Knox, G. A. 2000. The Ecology of Seashores. CRC Press LLC, Boca Raton, FL. 557 pp.

Larkum, W.D., Orth, R.J., Duarte, C.M., 2006. Seagrasses: Biology Ecology and Conservation. Springer, The Netherlands.

Lentfer, H. P. and A. J. Maier. 1995. Breeding ecology of the black oystercatcher in the Beardslee Island region of Glacier Bay National Park. In Proceedings of the Third Glacier Bay Science Symposium Proceedings of the Third Glacier Bay Science Symposium.

Lewis, J. 1996. Coastal benthos and global warming: strategies and problems. Marine Pollution Bulletin 32:698-700.

Lewis, T. L., D. Esler, and W. S. Boyd. 2007. Effects of predation by sea ducks on clam abundance in soft-sediment intertidal habitats. Marine Ecology Progress Series 329:131-144.

Liddle, M. J. 1975. A selective review of the ecological effects of human trampling on natural ecosystems. Biological Conservation 7: 17-36.

Lindberg, D. R., K. I. Warheit and J. A. Estes. 1987. Prey Preference and Seasonal Predation by Oystercatchers on Limpets at San Nicolas Island, California USA. Marine Ecology. 39: 105-113.

Lindberg, D. R., J. A. Estes and K. I. Warheit. 1998. Human influences on trophic cascades along rocky shores. Ecological Applications 8(3): 880-890.

Marsh, C. P. 1986a. The Impact of Avian Predators on high intertidal limpet populations. J. Exp. Mar. Biol. Ecol. 104: 185-201.

Marsh, C. P. 1986b. Rocky Intertidal Community Organization: The Impact of Avian Predators on Mussel Recruitment. Ecology 67(3): 771-786.

McConnaughey, T. & McRoy, C. P. 1979. C13 label identifies eelgrass (Zostera marina L.)

carbon in an Alaskan estuarine food web. Mar. Biol. 53, 263-269.

McRoy, C.P. 1968. The distribution and biography of Zostera marina L. (eelgrass) in Alaska. Pac. Sci. 22, 507-513.

McRoy, C. P. 1970. Standing stocks and other features of eelgrass (Zostera marina L.) populations on the coast of Alaska. J. Fish. Res. Bd. Can. 27, 1811-1821.

Meese, R. J. 1990. Effects of Predation by Birds on Rocky Intertidal Community Structure. Bull. Ecol. Soc. Am. 71 (2): 250.

Meire, P. M. 1993. The Impact of Bird Predation on Marine and Estuarine Bivalve Populations: A selective Review of Patterns and Underlying Causes Bivalve Filter Feeders in Estuary. In Bivalve Filter Feeders in Estuarine and Coastal Ecosystem Processes. Vol. G33 Heidelberg, Germany: Springer Verlag.

Menge, B. A. and G. M. Branch. 2001. Rocky intertidal communities. in Marine Community Ecology M. D. G. S. D. H. M. E. Bertness, Brown University, University of California at Santa Barbara, Georgia Institute of Technology. Marine Community Ecology. Sinauer Associates, Inc., Sunderland, Massachusetts.

Monson, D. H., D. F. Doak, B. E. Ballachey, A. M. Johnson, and J. L. Bodkin. 2000. Long-term impacts of the Exxon Valdez oil spill on sea otters assessed through age-dependent mortality patterns. Proceedings of the National Academy of Sciences 97:6562-6567.

Neckles, H.A., Short, F.T., Barker, S., Kopp, B.S., 2005. Disturbance of eelgrass *Zostera marina* by commercial mussel *Mytilus edulis* harvesting in Maine: dragging impacts and habitat recovery. Mar. Ecol.-Prog. Ser. 285, 57–73.

O'Clair, C.E. and S. D. Rice. 1985. Depression of feeding and growth rates of the seastar Evasterias troschelii during long-term exposure to the water-soluble fraction of crude oil. Marine Biology 84: 331-340.

O'Clair, R.M and C.E. O'Clair. 1988. Southeast Alaska Rocky Shores: Animals. Plant Press, Auke Bay, Alaska.

Olesen, B., 1996. Regulation of light attenuation and eelgrass *Zostera marina* depth distribution in a Danish embayment. Mar. Ecol.-Prog. Ser. 134, 187–194.

Peterson, C. H. 2001. The "Exxon Valdez" oil spill in Alaska: acute, indirect and chronic effects on the ecosystem. Advances in Marine Biology 39: 1-103.

Peterson, C. H., S. D. Rice, J. W. Short, D. Esler, J. L. Bodkin, B. E. Ballachey, and D. B. Irons. 2003. Long-term ecosystem response to the Exxon Valdez oil spill. Science 302: 2082-2086.

Povey, A. and M. J. Keough. 1991. Effects of trampling on plant and animal populations on rocky shores. Oikos 61(3): 355-368.

Power, M. E., D. Tilman, J. A. Estes, B. A. Menge, W. J. Bond, L. S. Mills, G. Daily, J. C. Castilla, J. Lubchenco, and R. T. Paine. 1996. Challenges in the quest for keystones.

BioScience 46(8): 609-620.

Riedman, M. L. and J. A. Estes. 1990. The Sea Otter (Enhydra lutris): Behavior, Ecology, and Natural History. US Fish and Wildlife, National Technical Service. 90 (14).

Sagarin R. D., Barry J. P., Gilman S. E., and Baxter C. H. 1999. Climate related changes in an intertidal community over short and long time scales. Ecological Monographs 69(4): 465-490.

Schiel, D. R., and D. I. Taylor. 1999. Effects of trampling on a rocky intertidal algal assemblage in southern New Zealand. J. Exp. Mar. Biol. & Ecol. 235(2): 213-235Schultz, S.T. 2008. Seagrass monitoring by underwater videography: Disturbance regimes, sampling design, and statistical power. Aquat. Bot. 88: 228-238.

Short, F. T., and S. Wyllie-Echeverria. 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation 23(1): 17-27.

Short, F.T., Koch, E.W., Creed, J.C., Magalhaes, K.M., Fernandez, E., Gaeckle, J.L., 2006. Seagrass net monitoring across the Americas: case studies of seagrass decline. Mar. Ecol. 27, 277–289.

Sousa, W. P. 1979. Experimental Investigations of Disturbance and Ecological Succession in a Rocky Intertidal Algal Community. Ecological Monographs. 49(3): 227-254.

Stevens, D.L. and A.R. Olsen. 2004. Spatially Balanced Sampling of Natural Resources. Journal of the American Statistical Association 99:262-278.

Terrados, J., Grau-Castella, M., Pinol-Santina, D., Riera-Fernandez, P., 2006. Biomass and primary production of a 8–11 m depth meadow versus 3 m depth meadows of the seagrass *Cymodocea nodosa* (Ucria) Ascherson. Aquat. Bot. 84, 324–332.

Thayer, J. K. & Phillips, R. C. 1977. Importance of eelgrass beds in Puget Sound. Mar. Fish. Rev. 36, 27-39.

Tinker, M.T., D.F. Doak, J.A. Estes, B.B. Hatfield, M.M. Staedler, and J.L. Bodkin. 2006. Incorporating diverse data and realistic complexity into demographic estimation procedures for sea otters. Ecological Applications 16(6):2293-2312.

Tinker, M.T., G. Bentall, and J.A. Estes. 2008. Food limitation leads to behavioral diversification and dietary specialization in sea otters. Proceedings of the National Academy of Sciences 105(2)560-565.

Trust, K. A., D. Esler, B. R. Woodin and J. J. Stegeman. 2000. Cytochrome P450 1A induction in sea ducks inhabiting nearshore areas of Prince William Sound, Alaska. Marine Pollution Bulletin 40(5): 397-403.

Walker, D. L., Lukatelich, R. J., Bastyan, G. & McComb, A. J. 1989. Effect of boat moorings on seagrass beds around Perth, Western Australia. Aquat. Bot. 36, 69-77.

Webster, J. D. 1941. Feeding habitats of the black oyster-catcher. The Condor 43: 175-180.

Wootton, J. T. 1992. Indirect Effects, Prey Susceptibility, and Habitat Selection: Impacts of Birds on Limpets and Algae. Ecology 73(3): 981-991.

VanBlaricom., G.R. 1988. Effects of foraging by sea otters on mussel-dominated intertidal communities. Pages 48-91 *in* G.R. VanBlaricom and J.A. Estes (eds.) The Community Ecology of Sea Otters. Springer Verlag, Berlin, Germany.

VanBlaricom, G.R. and J.A. Estes, eds. 1988. The Community Ecology of Sea Otters. New York, NY. Springer-Verlag.

Vermeer, K., K. H. Morgan and G. E. J. Smith. 1992. Black oystercatcher habitat selection, reproductive success, and their relationship with Glaucous-winged gulls. Colonial Waterbirds 15(1): 14-23.

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

NPS 965/110809, October 2011

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science
1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525

www.nature.nps.gov